WATER QUALITY STUDIES: RICHARD B. RUSSELL AND CLARKS HILL LAKES

First Annual Interim Report

by

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This report, the first of three annual reports, documents the results of comprehensive water quality studies at Richard B. Russell and Clarks Hill Lakes, Georgia-South Carolina. The objectives of the studies were to document water quality conditions in each lake before, during, and following the impoundment of Richard B. Russell Lake and to evaluate the effectiveness of an oxygenation system installed in Richard B. Russell Lake.
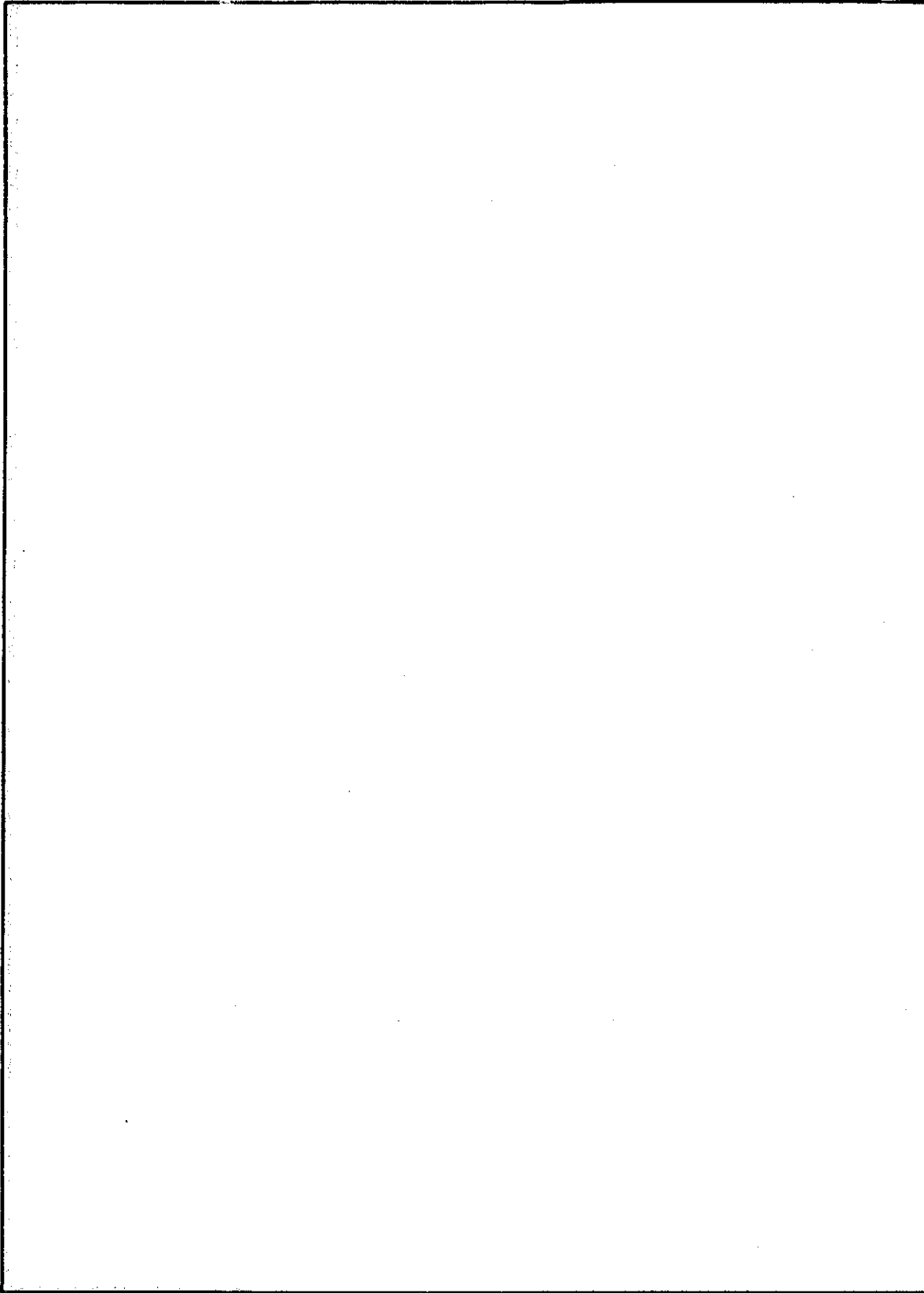
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EXECUTIVE SUMMARY

This report, which is the first of three annual reports documenting the results of comprehensive water quality studies at Richard B. Russell and Clarks Hill Lakes, pertains to the period October 1983 to December 1984. Presented in this report are detailed summaries of water quality conditions in both lakes during and immediately following impoundment of Richard B. Russell Lake.

In general, Richard B. Russell Lake exhibited impaired water quality conditions during the first year of impoundment. This trend appeared to be related to the inundation of highly labile organic material (i.e., vegetation and detritus) and operational discharges. The establishment of thermal stratification led to extensive hypolimnetic anoxia in a major portion of the reservoir and elevated concentrations of dissolved nutrients and metals in the bottom waters. The severity of these conditions was related, in part, to the effects of near-surface tainter gate releases which isolated the hypolimnion from exchanges with surface waters and prevented flushing. As a result, anoxic conditions were evident from the bottom to the 10-m depth by September. Exchanges of readily oxidizable materials (i.e., iron, manganese, and organic carbon) occurred between the epilimnion and hypolimnion during fall mixing. This resulted in severely depressed dissolved oxygen concentrations in the surface waters from November until December 1984, since these materials exerted a significant demand on dissolved oxygen which could not be immediately met by reaeration.

The impoundment of Richard B. Russell Lake had a minimal impact on the water quality of the headwater region of Clarks Hill Lake during the study period. Although Russell Lake exhibited severe water quality conditions in the hypolimnion during thermal stratification, discharges of surface water via tainter gates provided well-oxygenated water to Clarks Hill Lake. Overall, the headwater region of Clarks Hill Lake exhibited high dissolved oxygen concentrations (i.e., >6 mg/l) and acceptable chemical concentrations throughout the study period. The forebay area exhibited hypolimnetic anoxia late in the stratified period, and only moderate chemical changes were observed. These water quality patterns contrasted markedly with those observed in Richard B. Russell Lake.

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PREFACE

A 3-year, comprehensive water quality study at Richard B. Russell and Clarks Hill Lakes was initiated in October 1983 as a cooperative effort by the US Army Engineer District, Savannah, and the US Army Engineer Waterways Experiment Station (WES). This report, which covers the period October 1983 to December 1984, is the first of two annual interim reports documenting findings and results. A final report is to be prepared by April 1987. This report is submitted in accordance with the "Scope of Work: Water Quality Monitoring Program - Richard B. Russell Dam and Lake, Georgia and South Carolina" (Intra-Army Order No. PD-EI-84-07).

Portions of these studies were sponsored by the Office, Chief of Engineers (OCE), US Army, through the Water Operations Technical Support (WOTS) Program. WOTS Program Manager was Dr. J. L. Mahloch, WES.

This report was prepared by Mr. William F. James, Dr. Robert H. Kennedy, Dr. Stephen P. Schreiner, Mr. Steven Ashby, and Mr. Joe H. Carroll of the Environmental Laboratory (EL), WES. Participating in the conduct of the studies were Mr. Harry Eakin, Mr. William Jabour, Mr. Robert C. Gunkel, Jr., Dr. John Hains, and Dr. Robert F. Gaugush (EL). Dr. Kennedy and Mr. Carroll were responsible for the conduct of the studies and preparation of the report. The report was prepared under the direct supervision of Dr. Thomas L. Hart, Chief, Aquatic Processes and Effects Group, and under the general supervision of Mr. Donald L. Robey, Chief, Ecosystem Research and Simulation Division, and Dr. John Harrison, Chief, EL. The report was edited by Ms. Jessica S. Ruff of the WES Publications and Graphic Arts Division.

Director of WES was COL Allen F. Grum, USA. Technical Director was Dr. Robert W. Whalin.

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WATER QUALITY STUDIES: RICHARD B. RUSSELL AND CLARKS HILL LAKES

First Annual Interim Report

PART I: INTRODUCTION

Reservoirs provide an environmental resource of ever-increasing value. While historically constructed and operated primarily for flood control, hydroelectric power generation, navigation, and water supply, reservoir uses have, in recent years, been expanded to include fish and wildlife habitat, water quality control, and water-based recreation. This increased emphasis on recreational and environmental value coincides with a progressive decline in the quality of natural lakes located near population centers and an increased public demand for the protection of such resources. This, in turn, has prompted efforts to better understand reservoir water quality processes and to develop sound management strategies for the protection of this resource.

Much of our present understanding of water quality processes is based on the study of small, natural lakes. However, recent comparative evaluations of reservoir and natural lake characteristics (Thornton et al., 1981) indicate that significant differences exist between these two types of aquatic systems. These differences include greater size and morphologic complexity, higher material loadings, and a greater importance of advective transport for reservoirs than for natural lakes. Differences in the degree to which spatial patterns in water quality are exhibited are also apparent (Thornton et al., 1981; Kennedy et al., 1982). Reservoirs, particularly those with a long, narrow morphology, frequently display marked gradients in water quality characteristics along their major axis.

Perhaps most striking are differences in operation and origin. Since they are designed and operated to control flow, reservoirs and their discharges are highly controlled. Discharges from reservoirs also commonly occur from one or more depths in the water column. Natural lakes, on the other hand, have surface-overflow discharges and are regulated by local hydrologic conditions. Reservoirs are new geological features and, thus, morphologic characteristics are shaped by the topography of the preimpoundment landscape.

Of particular interest to limnologists and others concerned with the

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management of reservoir water quality are the extraordinary events which occur
in the years immediately following impoundment. This is a period of radical
change about which little is known. Among the obvious changes are significant
reductions in flow, increases in the rate at which sedimentation occurs, and
the inundation of terrestrial habitat. The consequences of these physical
changes, while poorly understood or documented, are potentially great. The
flooding of soils results in the immediate leaching of readily solubilized
metals, nutrients, and organic compounds. Continued interactions between
flooded soils and the overlying water column are also of potential signifi-
cance for a period of time following inundation due to changing chemical con-
ditions in these saturated soils. The decomposition of terrestrial vegetation
and detritus left in place prior to reservoir filling may donate significant
quantities of nutrients and organic compounds to the water column and exert
demands on dissolved oxygen stores. Under stratified conditions, excessive
oxygen demands lead to total oxygen depletion and the establishment of anoxic
conditions in bottom waters. This, in turn, accelerates the rate at which
materials are reduced and released at the sediment/water interface.

The net effect of such occurrences is the development of adverse and
often severe water quality conditions, both in the newly flooded pool and in
downstream areas, during the early years of impoundment. Commonly encountered
water quality management concerns are hypolimnetic anoxia, the accumulation
in bottom waters of reduced metals such as iron and manganese, and excessive
rates of algal production. In cases in which operational procedures require
the withdrawal of water from anoxic bottom strata, downstream areas are ex-
posed to elevated metal, nutrient, and hydrogen sulfide levels, and the influx
of oxygen-deficient release waters. Such conditions impact aesthetic, recrea-
tional, and water-use values and threaten fish and wildlife habitat.

Richard B. Russell Dam, authorized as Trotters Shoals Dam on 7 November
1966 by the "Flood Control Act of 1966," Public Law 89-789, Eighty-Ninth Con-
gress HR 18233, provides power generation, incidental flood control, recrea-
tion, streamflow regulation, and water supply. The reservoir, which is the
third major water control and recreational facility constructed on the Savan-
nah River by the US Army Corps of Engineers, is situated between Hartwell and
Clarks Hill Lakes.

Concerns over the potential environmental and water quality conditions
within the newly filled pool and in areas of Clarks Hill Lake immediately

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below Richard B. Russell Dam led to the development of a comprehensive miti-
gation plan for this reach of the Savannah River. To meet the water quality
quidelines set forth in this plan, the US Army Engineer District (USAED),
Savannah, developed a management approach involving the construction and
operation of an oxygen injection system and initiated a 3-year water quality
investigation at Richard B. Russell and Clarks Hill Lakes.

The major objectives to be addressed in the course of this investigation
are:

(1) To describe postimpoundment water quality conditions in Richard B.
    Russell Lake.

(2) To document the impacts of impoundment on water quality conditions
    in Clarks Hill Lake.

(3) To evaluate the effectiveness of the oxygen injection system in
    ameliorating potential water quality problems in Richard B. Russell
    Lake and its tailwater.

To accomplish these objectives, the USAED, Savannah, entered into a coopera-
tive agreement with the US Army Engineer Waterways Experiment Station (WES).
As a result of this agreement, a water quality laboratory was established
onsite, and water quality monitoring activities began in October 1983. The
scheduled duration of these studies is 3 years with a final report of findings
to be prepared by April 1987.

Concerns of particular interest during the first year of the investiga-
tion centered on evaluations of the immediate impacts of impoundment. These
included patterns in thermal stratification, dissolved oxygen depletion, and
nutrient and metal dynamics. Emphasis was also placed on the identification
of interactions between releases and headwater areas of both lakes, and on
the delineation of baseline water quality conditions in Clarks Hill Lake.

Study approaches involved a combination of routine and event- or
process-oriented data collection efforts in both Richard B. Russell and Clarks
Hill Lakes. Representative sampling stations were established in both lakes
and their tailwaters for monitoring purposes. Event- or process-oriented
studies, which were designed to more completely delineate specific events or
processes influencing water quality, were generally conducted over short pe-
riods of time and with greater sampling effort. Data collected through the
application of these varied study approaches will provide a detailed under-
standing of water quality conditions and processes, and the necessary infor-
mational basis for the development of sound management guidelines.

PART II: STUDY SITE HISTORY AND DESCRIPTION

Introduction

The study area consists of a 160-km section of the Savannah River extending along the Georgia and South Carolina border and includes three major US Army Corps of Engineer impoundments: Clarks Hill, Richard B. Russell, and Hartwell Lakes (Figure 1). Clarks Hill, Richard B. Russell, and Hartwell Dams are located at river kilometers 386, 442, and 488, respectively. All three lakes are multipurpose projects designed and operated primarily for hydropower production, flood control, water supply, recreation, and navigation.

Watershed Description

The Savannah River and its basin drain approximately 27,400 sq km and reach from the river's headwaters in the Blue Ridge Mountains of southwestern North Carolina to the Atlantic coast near Savannah, Ga. More than half of the drainage area, some 16,000 sq km, is located above the Clarks Hill Dam. The maximum length of the drainage basin is approximately 400 km; maximum width is approximately 113 km.

The Savannah River is formed by the confluence of the Seneca and Tugaloo Rivers, both of which have their headwaters on the southern slope of the Blue Ridge Mountains in North Carolina. The river meanders in a southeasterly direction through the Piedmont Plateau and Coastal Plain, and forms the boundary between Georgia and South Carolina.

Topography in the study area is characterized by gently sloping upland areas cut by gullies and stream valleys; relief becomes more rugged in areas adjacent to the Savannah River where well-developed, moderately steep to steep ridges and ravines form the topography. Elevation varies within the study area from a maximum of 274 m mean sea level (MSL) in the Hartwell Lake area to 55 m MSL in the Clarks Hill Dam release area.

The drainage basin is located in the Piedmont Physiographic Province and is underlain by Precambrian-age granite and metasedimentary gneiss and schist of the Little River series. The stratigraphic units of the area consist of rocks of the Lower Little River series, recent alluvial soils occurring as silty and poorly graded sands high in organic material, and residual soils
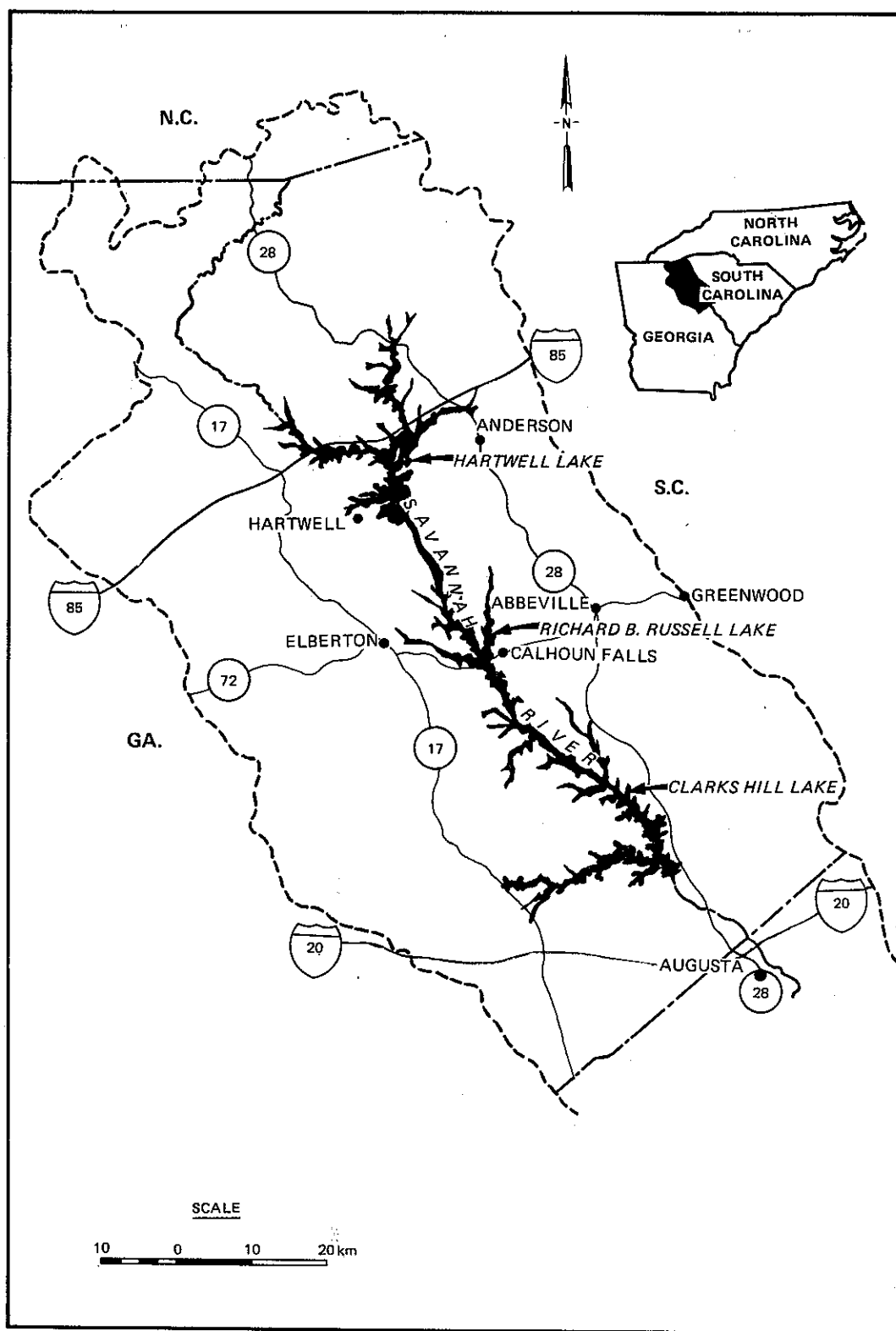
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Figure 1. Savannah River watershed indicating the locations of Hartwell, Richard B. Russell, and Clarks Hill Lakes

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consisting of mixtures of sand, silt, and clay. The rocks are of igneous or metamorphic origin and include granites, gneisses, schists, basic eruptives, and highly metamorphosed shales, sandstones, and limestones. Those of igneous origin usually contain ferro-magnesium and feldspar-like minerals.

Land use is fairly uniform throughout the study area with woodlands comprising 60 percent and pasture and cropland about 35 percent of the area; only a small portion of the watershed is exposed to urban and recreational uses. The wooded uplands consist of mixed pine and hardwood forest, and lumber harvesting is one of the major industries of the area. Cleared lands are devoted primarily to cultivated crops and pasture. Principal crops grown are cotton, corn, hay, and grains. Cattle raising has become popular throughout the area. One other particularly important industry dependent upon the natural resources of the area is granite, which is quarried and shipped from Elbert County, Ga.

The waters of the Savannah River above Clarks Hill Dam are relatively free of pollution. Currently, point sources of pollution are minimal with most occurring in the Hartwell Lake area. However, numerous small waste water treatment plants, textile mills, and various other industries discharge wastes to streams and lakes within the study area. Due to the abundance of granitic rock, poorly drained soils, and the steepness of slopes, surface water tends to be low in dissolved solids and alkalinity and to have low buffering capacity.

Reservoir Characteristics

Clarks Hill Lake. Clarks Hill Dam was completed in 1954 and, although two smaller dams were already in place downstream, became the first major impoundment on the Savannah River. Clarks Hill Lake is a multipurpose reservoir providing flood control, hydroelectric power, recreation, fish and wildlife habitat, and flow regulation.

The dam is a combination concrete and earth-filled structure with penstocks and powerhouse located along the easternmost portion of the dam. The concrete section of the dam is 696 m long and rises 55 m above the streambed. The spillway is a concrete gravity ogee section extending across the west flood plain and river channel and consists of 23 spillway gates 12.2 m wide and 10.7 m high. The power intake section on the east side of the river channel contains seven 6.1-m-diam low-level penstocks spaced 18.9 m on center.

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The powerhouse consists of seven generators with a total rated capacity of 280,000 kW.

Clarks Hill Dam is located just downstream of the confluence of the Savannah and Little Rivers, the inundated valleys of which create the two major arms of the lake. The Little River arm extends 50 km west of the dam into Georgia, while the Savannah River arm extends 63 km northwest to the Richard B. Russell Dam. Both arms of the lake are long, narrow, and dendritic. A shoreline development ratio of 32 indicates the extreme irregularity of the 1,930-km shoreline. At maximum power pool elevation (100.6 m MSL), the reservoir has a surface area of 283.2 sq km and a volume of 3.023 billion cu m. Average discharge from the Clarks Hill Dam is 243.6 cubic meters per second (cms). The theoretical hydraulic retention time at maximum power pool is 144 days.

Normal releases from Clarks Hill Lake occur through the powerhouse; however, flood discharges may be passed through spillway gates. The lake is generally near full power pool from May through August. From September through November the pool is drawn down to an elevation of 97.6-99.1 m MSL. Inflows to the lake are highest in December, and increases in pool elevation are reported through March.

Clarks Hill Lake is a warm monomictic reservoir exhibiting strong thermal stratification, low nutrient and dissolved solids concentrations, and low alkalinity. Algal biomass is generally low, and algal blooms are infrequent. Bloom conditions, when they do exist, are generally localized in the secondary tributary arms where higher nutrient levels are observed.

The onset of stratification typically occurs in April or May, and fall mixing is initiated in October. Mixed conditions prevail from November through March. During periods of stratification, hypolimnetic anoxia develops from the near-dam area to near midpool. During such periods, higher inorganic carbon, iron, manganese, and sulfide concentrations are observed in the bottom water. The extent and duration of anoxia have, however, decreased in recent years relative to conditions during the years immediately following initial impoundment of the lake. Release water quality is relatively good except during periods when anoxic hypolimnetic waters exist near the dam and in the withdrawal zone of the penstocks. Other limnological and physical characteristics of the lake are presented in Tables 1 and 2.

Richard B. Russell Lake. The impoundment of the Savannah River by

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Richard B. Russell Dam occurred in 1983. Filling operations began in December 1983, and maximum power pool (145 m MSL) was reached in November 1984. Work on the dam will continue until completion of the powerhouse in late 1989. Russell Dam is the third multipurpose US Army Corps of Engineers project built on the Savannah River and is located 63 km upstream of Clarks Hill Dam and 48 km downstream from Hartwell Dam. Russell Dam impounds all but approximately 1 km of the Savannah River between Clarks Hill and Hartwell Lakes. Project purposes include power production, incidental flood control, recreation, streamflow regulation, and water supply.

Two other major tributaries impounded by the Russell Dam are Rocky River and Beaverdam Creek. The confluence of both of these tributaries with the Savannah River was located 8 km upstream of the present location of the Russell Dam. Rocky River extends 20 km into Abbeville County, S. C., north of the Savannah River, while Beaverdam Creek drains a 17-km-long watershed to the west and northwest in Elbert County, Ga. The area drained by these and other secondary tributaries, and by direct runoff to Richard B. Russell Lake, is approximately 2,100 sq km. This compares to a 5,406-sq km drainage area for the Savannah River above the headwaters of Richard B. Russell Lake.

Richard B. Russell Lake is large (surface area, 107.9 sq km; volume, 1.271 billion cu m) and morphologically complex (shoreline length, 885 km; shoreline development ratio, 24.5). Mean and maximum depths are 12 and 47 m, respectively. Discharges from Russell Dam will average 143.84 cms, and the theoretical hydraulic retention time will be 102 days. Other morphometric data are presented in Table 1.

Operation of Russell Dam will be similar to that of the Clarks Hill and Hartwell Dams in that power will be produced only at periods of peak daily demand, and releases for other purposes will be minimal. Russell Lake will maintain full power pool elevation (145 m MSL) for the entire year of operation. At this elevation, the lake will contain 173 million cu m of flood storage and 155 million cu m of usable storage between elevations 143 and 145 m MSL.

The Savannah River was strongly influenced by peaking power discharges from Hartwell Lake prior to the impoundment of Russell Lake. Discharges, occurring for periods of 4 to 6 hr in early to late afternoon on weekdays, produced rapid changes in flow, depth, temperature, dissolved oxygen, and other related chemical properties in the tailwater region of the Savannah River.

The time of travel to the Clarks Hill headwater region was approximately 8 hr. Midhypolimnetic releases from Hartwell Lake exhibited low concentrations of dissolved oxygen and poor water quality during thermal stratification. However, turbulent flow along the river stretch effectively reaerated and improved the quality of this water before it reached Clarks Hill Lake.

Richard B. Russell Dam is a concrete gravity structure flanked by earthen embankments. The powerhouse, with adjacent switchyard, is located immediately below the dam in the western floodplain. The concrete portion of the dam rises 59.5 m above the streambed and is 481.9 m long. The spillway, which is of the concrete gravity ogee type, is located over the old Savannah River streambed on the east side of the dam. The spillway consists of 10 tainter gates, each 15.3 m wide and 13.7 m high, and a flip bucket with a radius of 15.3 m. The spillway crest is at elevation 133 m MSL, and the flip bucket lip is at elevation 108.9 m MSL. The power intake section is on the west side of the dam and contains eight low-level, 7.9-m-diam penstocks. After completion, the powerhouse will contain eight generators rated at 75,000 kW each; four will be pumpback units. The powerhouse will eventually have a total rated capacity of 600,000 kW.

Hartwell Lake. Hartwell Lake was filled in 1962 and became the second multipurpose project completed as part of the comprehensive plan of development for the Savannah River basin. The Hartwell Dam lies 11 km below the confluence of the Tugaloo and Seneca Rivers, which form the Savannah River at that point. Hartwell Lake provides hydroelectric power, flood control, recreation, fish and wildlife habitat, and low-flow augmentaton.

Hartwell Dam consists of a concrete portion flanked by earthen embankments. The powerhouse and adjacent switchyard are located along the western floodplain immediately below the dam. The concrete portion of the dam rises 62.2 m above the streambed and is 579.5 m long. The spillway is a concrete gravity ogee section located in the river channel and consists of 12 spillway gates 12.2 m wide and 11.0 m high. The power intake section on the west end of the nonoverflow concrete section contains five low-level penstocks. The powerhouse generators have a total generating capacity of 344,000 kW.

The lake consists of two major basins: the Tugaloo arm, which is 79 km long and forms the Georgia-South Carolina state line northwest of the Hartwell Dam; and the Seneca arm, which extends 72 km north of the dam into South Carolina. Total drainage for Hartwell Lake is 5,406 sq km.

At maximum power pool elevation (201.2 m MSL), the reservoir has a sur-
face area of 226.6 sq km and a volume of 3.147 billion cu m (Table 1). Aver-
age discharge from the Hartwell Dam is 119 cms, and the theoretical hydraulic
retention time is 306 days at maximum power pool volume. Mean and maximum
depths are 14 and 55 m, respectively. Hartwell Lake has a shoreline length
of 1,548 km and a shoreline development ratio of 29.

Hartwell Dam produces power only during periods of maximum daily demand.
Low-flow augmentation requires that flow from at least one generator unit be
maintained for at least 1 hr during every 48-hr period for preservation of
fish habitat below the dam. This procedure is necessary only on off-peak
weekend periods.

Hartwell Lake is a warm, monomictic reservoir that exhibits strong ther-
mal stratification during summer. The lake is low in nutrients, turbidity,
solids, conductivity, alkalinity, and algal biomass. Patterns in thermal
stratification are similar to those observed in Clarks Hill Lake. Stratifi-
cation is evident from April until September. Although the severity of hypo-
limnetic oxygen depletion has decreased since impoundment, releases continue
to exhibit low dissolved oxygen concentrations (i.e., near 5 mg/l) late in
the stratified period. Other limnological characteristics are presented in
Table 2.

PART III: METHODS AND MATERIALS

Facilities and Personnel

The Richard B. Russell Limnological Laboratory was established by the WES in cooperation with the USAED, Savannah. Several advantages of this facility were anticipated. These included greater flexibility in meeting sampling needs, more rapid response to changing conditions, and the early incorporation of monitoring information in the operational decision-making process.

The facilities, which are located west of Calhoun Falls, S. C., on Highway 72, consist of a doublewide trailer housing a complete water quality laboratory, a singlewide trailer providing office space, and outbuildings for equipment storage and maintenance. The laboratory structure is partitioned into several small rooms for the conduct of selected analyses and a large, central area for sample processing, equipment calibration, and general use.

The office and laboratory facilities are situated on a peninsula approximately 0.25 mile from the shore where a boat docking and storage facility is located. Sampling craft used during the study period included a 24- and 26-ft (7- and 8-m) MonArk (MonArk, Monticello, Ark.), and a 15- and 18-ft (4.5- and 5.5-m) Boston Whaler (Boston Whaler Inc., Rockland, Mass.).

Personnel needs at the laboratory were met by four full-time and four temporary individuals who were responsible for all sampling, data collection and transfer, and laboratory analyses. Additional temporary employees were engaged during periods when intensive field data collection efforts were required. Personnel of the WES were involved in data base management, data interpretation, and overall project management.

Sampling Locations

Twenty primary stations were established from the tailwaters of Clarks Hill Lake to the tailwaters of Hartwell Lake (Figures 2 and 3). Most stations were located along the main longitudinal axis of Clarks Hill Lake (Stations 20, 30, 40, and 50) and Richard B. Russell Lake (Stations 60, 80, 100, 120, 160, 180, 190, 200). Stations were also located in the Beaverdam Creek Station 130), Rocky River (Stations 140 and 150), and Coldwater Creek (Station 170) embayments in Richard B. Russell Lake. Several stations were

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Figure 2.   Location of sampling stations on Clarks Hill Lake

HARTWELL LAKE

200

190

RICHARD B. RUSSELL LAKE

N

180

170

150

160

130

140

120

RUSSELL WATER QUALITY
LABORATORY

110

100

90

80

70

60

RICHARD B. RUSSELL DAM
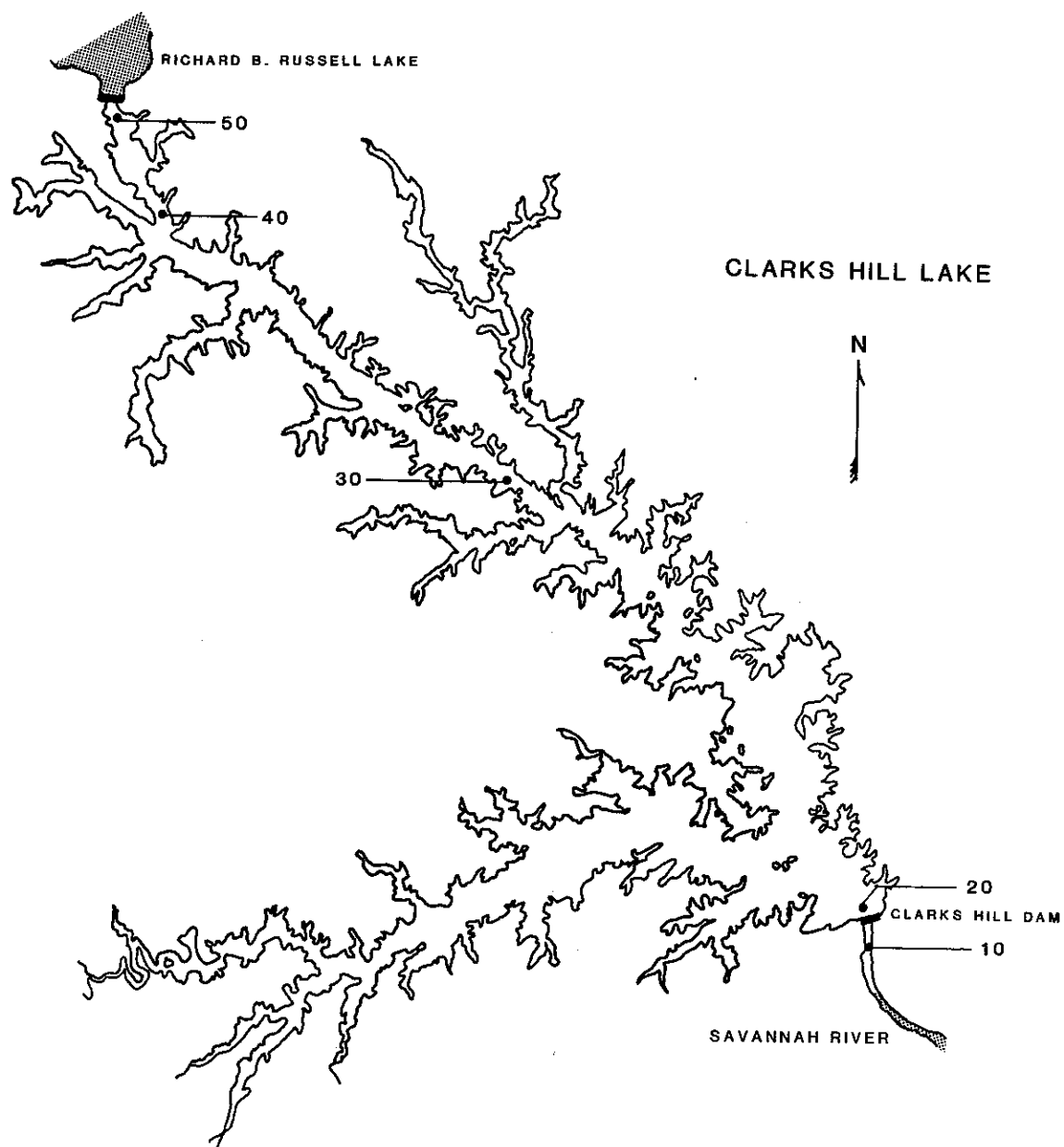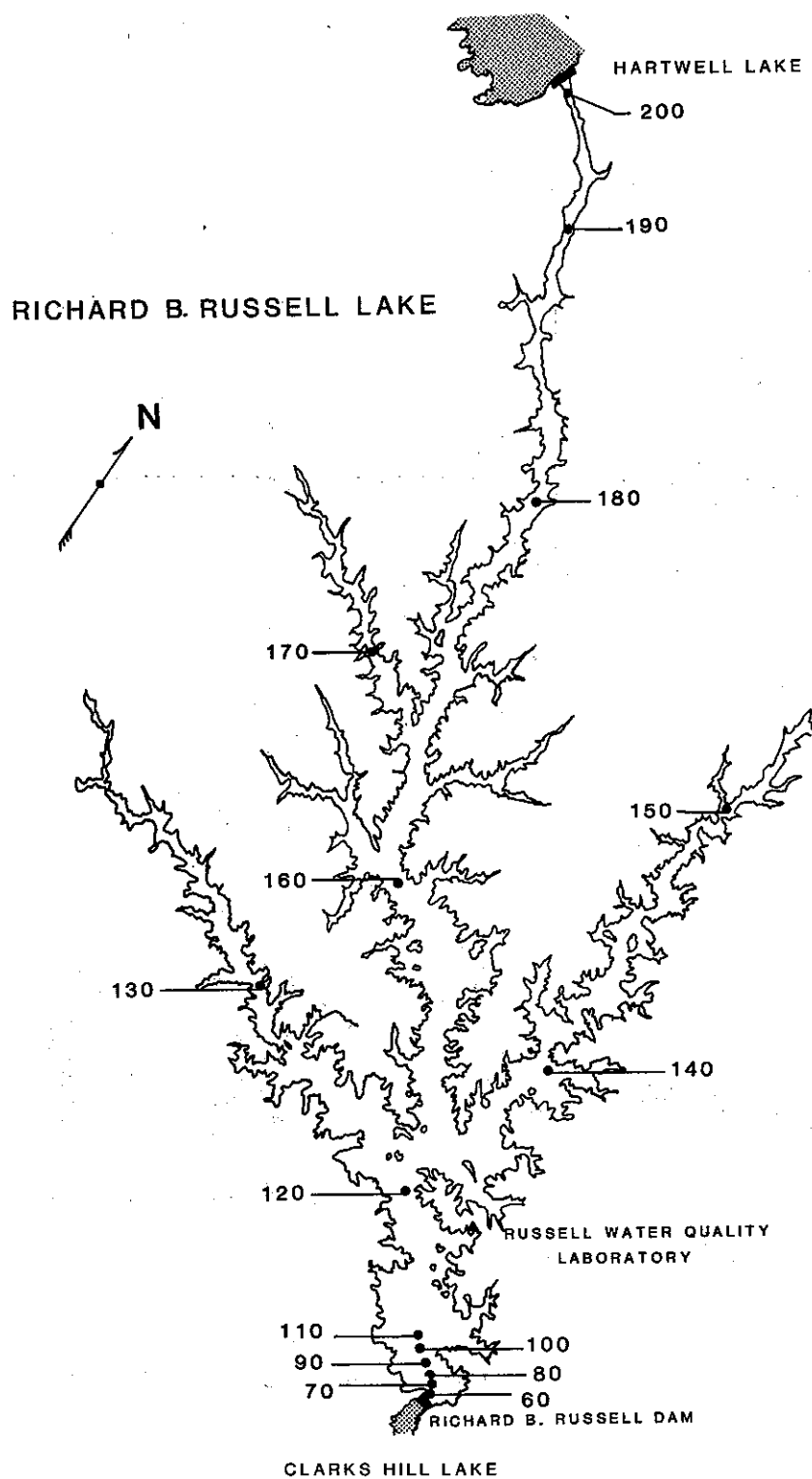
CLARKS HILL LAKE

Figure 3. Location of sampling stations on
Richard B. Russell Lake

concentrated in the vicinity of the oxygen injection system and the immediate downstream area of the lake (i.e., Stations 60-110) for evaluation of the system. These stations were located along three-station transects established perpendicular to the river channel. While several of these stations were routinely monitored for water quality, future operation of the oxygenation system will result in greater sampling efforts at these sites.

Sample depths were chosen so as to define vertical gradients in water quality. In-situ sampling for temperature, dissolved oxygen, pH, and specific conductance was conducted at 2-m intervals from top to bottom at all stations; 1-m intervals were used when significant vertical gradients were observed. The surface measurement was taken approximately 0.1 m below the surface, and the bottom measurement approximately 0.5 m from the bottom. In-situ measurements at Stations 10, 50, and 200 (i.e., tailwater stations) were taken at a depth of approximately 0.5 m. Station 60, because of its importance in describing conditions immediately upstream from the Richard B. Russell Dam, was profiled at 1-m intervals.

Water samples for chemical analyses were obtained from selected depths at each station. The selection of depths was based on vertical patterns in the distribution of in-situ variables and a desire to describe vertical differences in chemical conditions. During periods when the lake was well mixed (i.e., minimal thermal structure), a single sample was collected at middepth. These conditions were observed after fall mixing, during the winter months, and routinely at each of the outflow stations. When thermal structure existed or when significant dissolved oxygen gradients were observed, samples were obtained from three to six different depths; one to three samples were collected in the epilimnion (depending upon depth of the thermocline), one at midthermocline, and one to three in the hypolimnion. When the thermocline was well established, additional samples were collected 1 m above and below the midpoint of the thermocline. The uppermost water sample at each station was always collected 1 m below water surface while the bottommost sample was collected 0.5 to 1 m above bottom sediments. Supplemental stations and/or depths were added at various times as needed to better describe water quality conditions.

Sampling Schedule

Routine sampling consisted of monthly in-situ monitoring and seasonal

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physicochemical analyses (Table 3). The four seasonal sampling times coincided with the spring high-flow period, the early and late stratification periods, and the period following fall mixing. Analyses conducted on seasonally collected water samples are listed in Table 4.

In general, monthly in-situ data and seasonal physicochemical samples were collected at 17 of the 20 primary stations. Stations 70, 90, and 110 were to have been sampled only in conjunction with the operation of the oxygenation system.

In addition to the routine water quality monitoring, several intensive sampling efforts were conducted to describe process-oriented events. These included one dye study and two diel studies. These are described in Part IV of this report. Several unscheduled intensive sampling efforts were also conducted. These consisted of increases in the frequency of data collection at one or more selected stations and/or increases in the number of stations sampled. Such intensive efforts were deemed necessary during several periods of rapidly changing water quality conditions and in response to problem conditions or changes in operational procedures. Two such efforts of major significance involved attempts to monitor water quality conditions in the Rocky River and Beaverdam Creek embayments of Richard B. Russell Lake, and in the vicinity of the Richard B. Russell Dam during periods of generator testing.

Sampling Methods

In-situ variables included temperature, dissolved oxygen, pH, specific conductance, and oxidation-reduction potential. These measurements were made with either a Hydrolab Surveyor (Hydrolab Corp., Austin, Tex.) or a Martek Mark VIII (Martek Instruments Inc., Irvine, Calif.). Monitoring equipment was calibrated prior to field use by placing the sonde(s) in a large calibration tank; making independent measurements of temperature, dissolved oxygen, pH, and specific conductance; and adjusting the unit(s) to the nearest measurable value. Temperature was measured with an NBS thermometer to the nearest 0.1 deg. C; dissolved oxygen, by Winkler titration; and pH, electronically with a pH meter to the nearest 0.1 unit. Specific conductivity was determined with a Barnstead Wheatstone bridge. Transparency was determined using a standard 20-cm Secchi disc.

Measurement of in-situ variables was accomplished by lowering the sonde

to selected depths and recording response values from the instrument's deck unit after a period sufficiently long to ensure a stable meter reading. Replicate readings were obtained at selected depths as the sonde was retrieved to the surface. All readings and measurements, as well as the comments and observations of field personnel, were recorded on specially designed code forms. These forms were returned to the laboratory for review and direct entry to the computerized data base management system.

Field instruments were checked for calibration upon return to the laboratory following each sampling trip. Variances in readings, if observed, were noted and recorded. Significant variation from expected values required that the data be disregarded. However, this was rare in occurrence.

Sample collection for analysis of chemical and biological parameters involved discrete, grab or integrated samples, depending upon the station and parameter to be estimated. Discrete samples were collected at lake stations with a 12-V, diaphragm pump and vinyl-lined garden hose. Pumped samples were obtained by lowering the sampling hose to a desired depth, allowing the hose to clear by pumping a volume of water equivalent to two to three times the volume of the hose, and then retaining the necessary volume of water for the sample. Grab samples were collected with a polyethylene bucket at outflow stations.

Integrated samples for biological variables were collected with a 4-m-long, polyvinyl chloride pipe (3.8 cm, inside diameter) fitted at the lower end with a one-way check valve. Samples were collected by lowering the device to the desired depth (determined as twice the Secchi disc depth), allowing the valve to close during retrieval, and transferring the entrapped water to a sample bottle.

Analytical Methods

Analytical methods, digestion and filtration techniques, sample holding times, glassware preparation, reagents and chemicals used, and quality control are detailed in Appendix A. Accepted methods for analysis of water samples (i.e., AHPA, 1980; USEPA, 1974) were utilized in the laboratory whenever possible.

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PART IV: RESULTS AND DISCUSSION

Hydrologic Conditions

Changes in pool elevation, monthly precipitation, daily inflow, and daily discharges for Hartwell Lake for the period October 1983 to December 1984 are presented in Figure 4.   Pool elevation increased from a mean of 199.8 m MSL in October 1983 to a mean of 201.6 m MSL in December.   This was
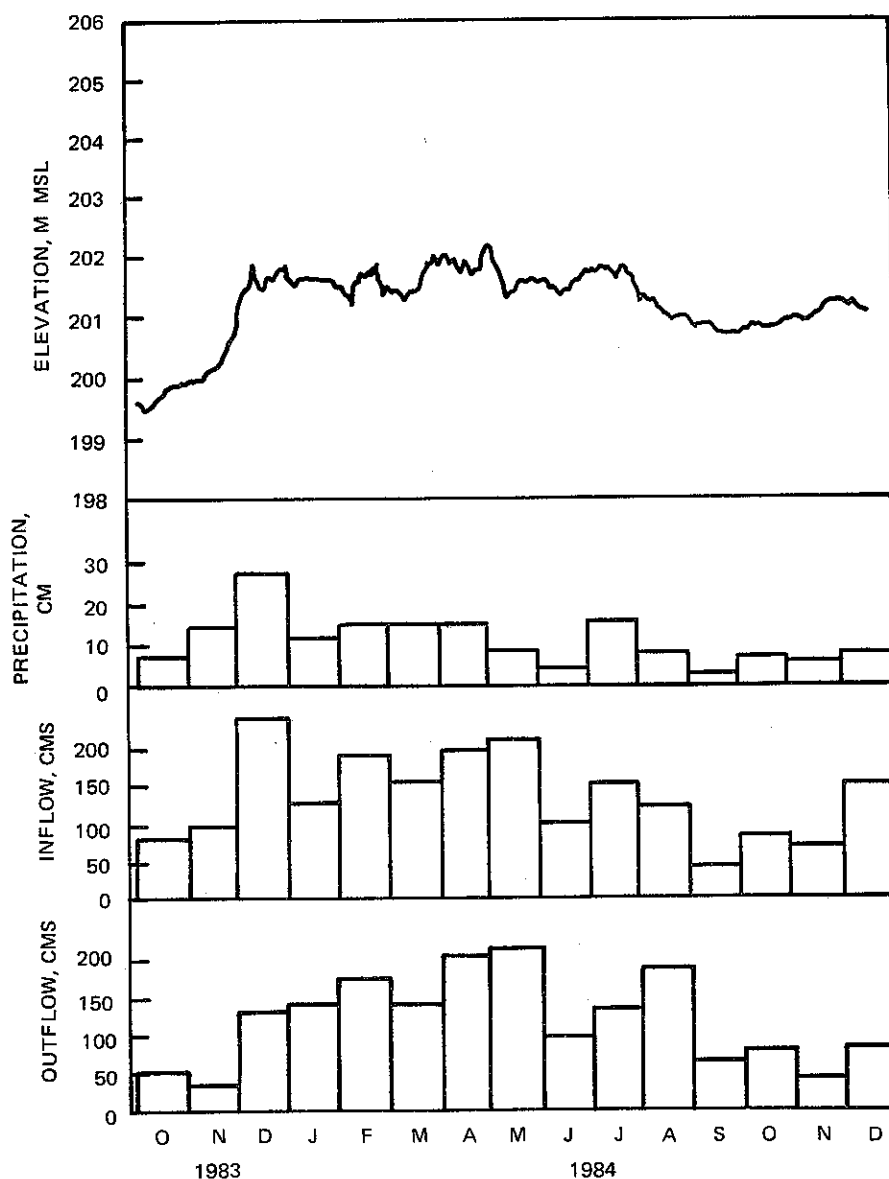

Figure 4.   Seasonal variations in pool elevation, monthly precipitation, mean daily inflow, and mean daily discharge for Hartwell Lake during 1983-1984

due to the regulation of discharges and a high daily inflow rate. Pool elevation exhibited minimal fluctuations from January until July, then declined slightly in October to 200.9 m MSL. Monthly precipitation and inflow rates were highest from December 1983 to May 1984, and lowest during the summer months.

Associated with higher water loads in the winter were elevated discharge rates. Monthly average discharge from Hartwell Dam increased from a low of 34 cms in November 1983 to 218.4 cms in May 1984. During the summer and fall period, average discharges were lower. Monthly discharges during this period were highest in August and lowest in November.

Clarks Hill Lake exhibited more pronounced fluctuations in pool elevation and higher annual rates of daily inflow and discharge than Hartwell Lake. While pool elevation was maintained at approximately 101 m MSL from December until July, elevation declined sharply between August and December (Figure 5).

Highest mean monthly precipitation rates were observed during the winter months in this area of the watershed; rates during the summer months were minimal. Daily inflow and discharge rates were also elevated in winter and lower in summer. However, Clarks Hill Lake exhibited higher mean daily inflow and discharge rates than those observed for Hartwell Lake.

The filling of Richard B. Russell Lake was marked by a series of pool elevation increases during the winter months and the maintenance of a stable pool elevation during the summer months (Figure 6). Abrupt changes in elevation and associated changes in inundated surface area had an indirect impact on water quality conditions in the lake during the first year of impoundment. During the filling period, three distinct phases can be identified. Phase I represents the period prior to significant rises in pool elevation (i.e., before October 1983). During this phase, pool elevation was maintained at approximately 105.2 m MSL and surface area was less than 500 ha (Figure 6).

Phase II of filling, which lasted from early October until early December 1983, was marked by a 7.5-m rise in pool elevation. Although minimal change in pool surface area occurred during this phase, the inundation of a breached coffer dike had a significant impact on flow patterns in the reservoir.

Closure of the sluice gates in December 1983 resulted in a rapid pool rise and a 19-fold increase in lake surface area. During this period, identified as Phase III, pool elevation increased from approximately 111.3 m MSL
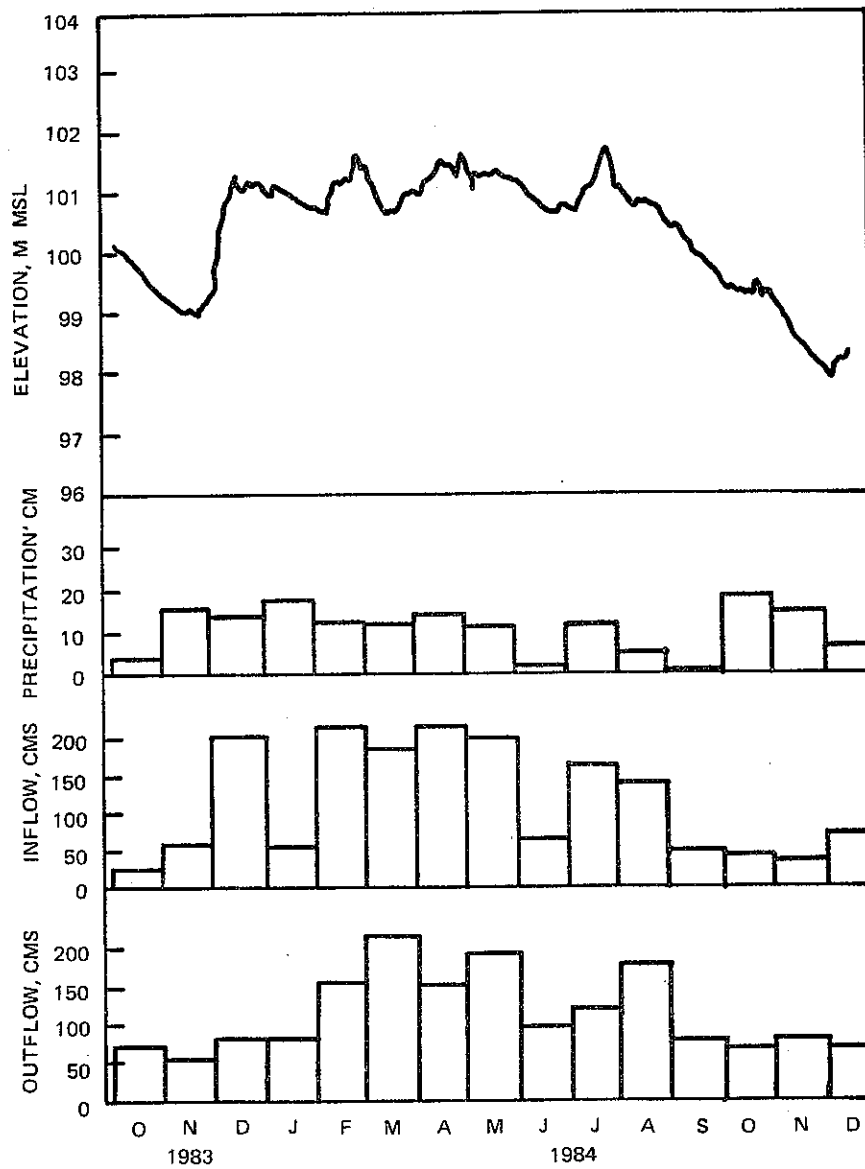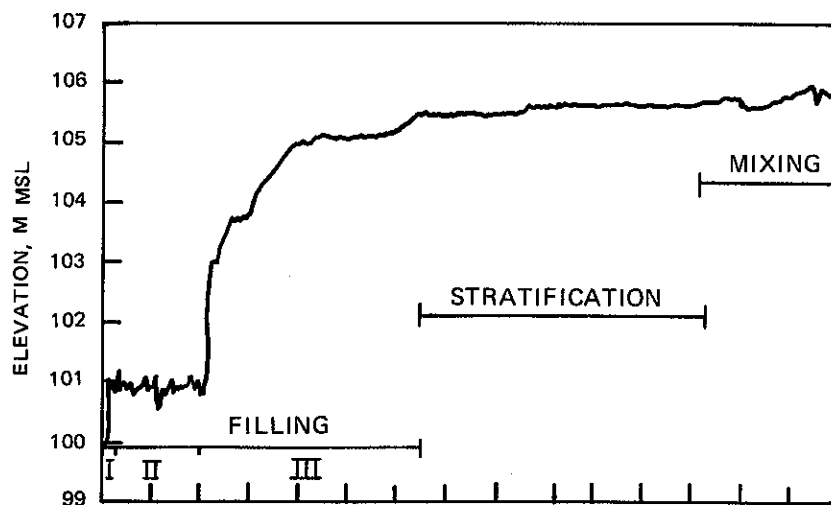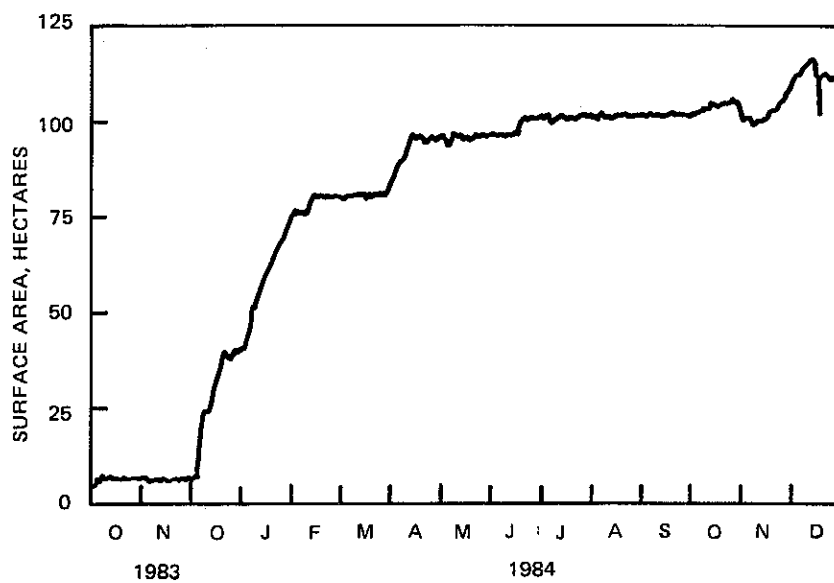
Figure 5.   Seasonal variations in pool elevation, monthly
precipitation, mean daily inflow, and mean daily discharge
for Clarks Hill Lake during 1983-1984

a.   Pool elevation

b.   Lake surface area

Figure 6.   Seasonal variations, Richard B. Russell
Lake, 1983-1984.   Bars indicate important opera-
tional or seasonal events

in December 1983 to 141.8 m MSL in April 1984. Because of tree-topping operations during this phase, a stable pool elevation was maintained from mid-February until April.

Pool elevation was maintained at 1.5 m below maximum power pool elevation (i.e., at approximately 143.4 m MSL) from April until November 1984, then allowed to rise to 144.9 m MSL in December. It was during this period that stratified conditions were observed. Lake surface at that time was approximately 10,500 ha.

Operational discharges during the first year of impoundment were primarily via bottom sluice gates and tainter gates. Releases occurred through sluice gates from October 1983 to April 1984. However, due to anticipated poor water quality conditions associated with a hypolimnetic withdrawal during summer stratification, operation was switched to tainter gates from May until December 1984. Testing of the first generator unit, which began in November 1984, resulted in the first releases from the penstocks. Because water quality conditions in the penstock withdrawal zone were poor during this period, near-surface releases from tainter gates were also conducted to improve the quality of the releases.

These hydrologic and operational changes provide a framework within which to discuss limnological patterns in Richard B. Russell Lake. In the discussion that follows, changing limnological conditions are identified during the filling period, the period of stable stratification, and the period of mixing.

Limnological Studies on Richard B. Russell Lake

Limnological conditions during the filling period. Limnological patterns during the filling period were related to changes in pool elevation, precipitation, hydraulic residence time, and an increasing area of inundated bottomlands. Events during this period occurred in three distinct phases. During Phase I (i.e., before filling), few spatial and temporal patterns were detected in temperature, dissolved oxygen, specific conductance, and pH. This was related to the lake's shallow mean depth, strong riverine currents, and a rapid flushing rate. Maximum depth at Station 60 was only 5 m, and surface inflows occurred only 3 km upstream from the dam. This promoted mixing and the persistence of riverine conditions. Temperature and dissolved oxygen

patterns suggested periods of slight stratification and intermittent mixing. On 3 October 1983, for instance, temperature ranged from 20 deg. C at the surface to 17 deg. C at the bottom of Station 60, and dissolved oxygen concentrations were high (7-9 mg/l) and uniform throughout the water column.

Phase II of filling was marked by a pool rise of approximately 7.5 m in mid-October. During this phase, water quality patterns reflected the effects of a submerged coffer dike on flow patterns in Richard B. Russell Lake. The breached coffer dike, which spanned the thalweg with a crest elevation of 345 m MSL was submerged at a depth of approximately 6 m. Data collected on 19 October documented the upward deflection of temperature isotherms in the area of the coffer dike and existence of distinct patterns in the distribution of dissolved oxygen and specific conductance (Figure 7). Also apparent were the effects of discharges from the bottom sluice gates. The coffer dike appeared to be acting as a submerged weir allowing for the preferential discharge of surface waters and partial impoundment of bottom waters. This, in turn, led to periodic lowered dissolved oxygen concentrations and slightly elevated specific conductance values upstream of the coffer dike. These patterns were observed until further pool elevation increases and the operation of the tainter gates resulted in a change in flow characteristics in the forebay area.

Increases in pool length and depth during Phase II of filling also led to the establishment of longitudinal variations in water quality. For example, on 22 November 1983, specific conductance values exhibited a distinct vertical and longitudinal pattern near the point of inflow, suggestive of the occurrence of vertical stratification and interflowing density currents (Figure 8).

Increases in elevation during Phase III of filling (i.e., from December through April) resulted in greater influences from the Beaverdam Creek and Rocky River embayments. During this period, pool elevation increased from 110 to 140 m MSL. In-situ patterns in the main basin indicated nearly isothermal conditions, high dissolved oxygen concentrations, and uniform specific conductance values; however, marked differences in specific conductance values in the two embayments and conditions near bottom at Station 120, particularly following storm events, indicated the significance of inflows from these tributaries. This pattern was evident from data collected on 22 February 1984 (Figure 9).

Figure 7. Vertical and longitudinal patterns in temperature (upper panel), dissolved oxygen (middle panel), and specific conductance (lower panel) in Richard B. Russell Lake on 19 October 1983

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Figure 8. Vertical and longitudinal patterns in tempera-
ture (upper panel), dissolved oxygen (middle panel), and
specific conductance (lower panel) in Richard B. Russell
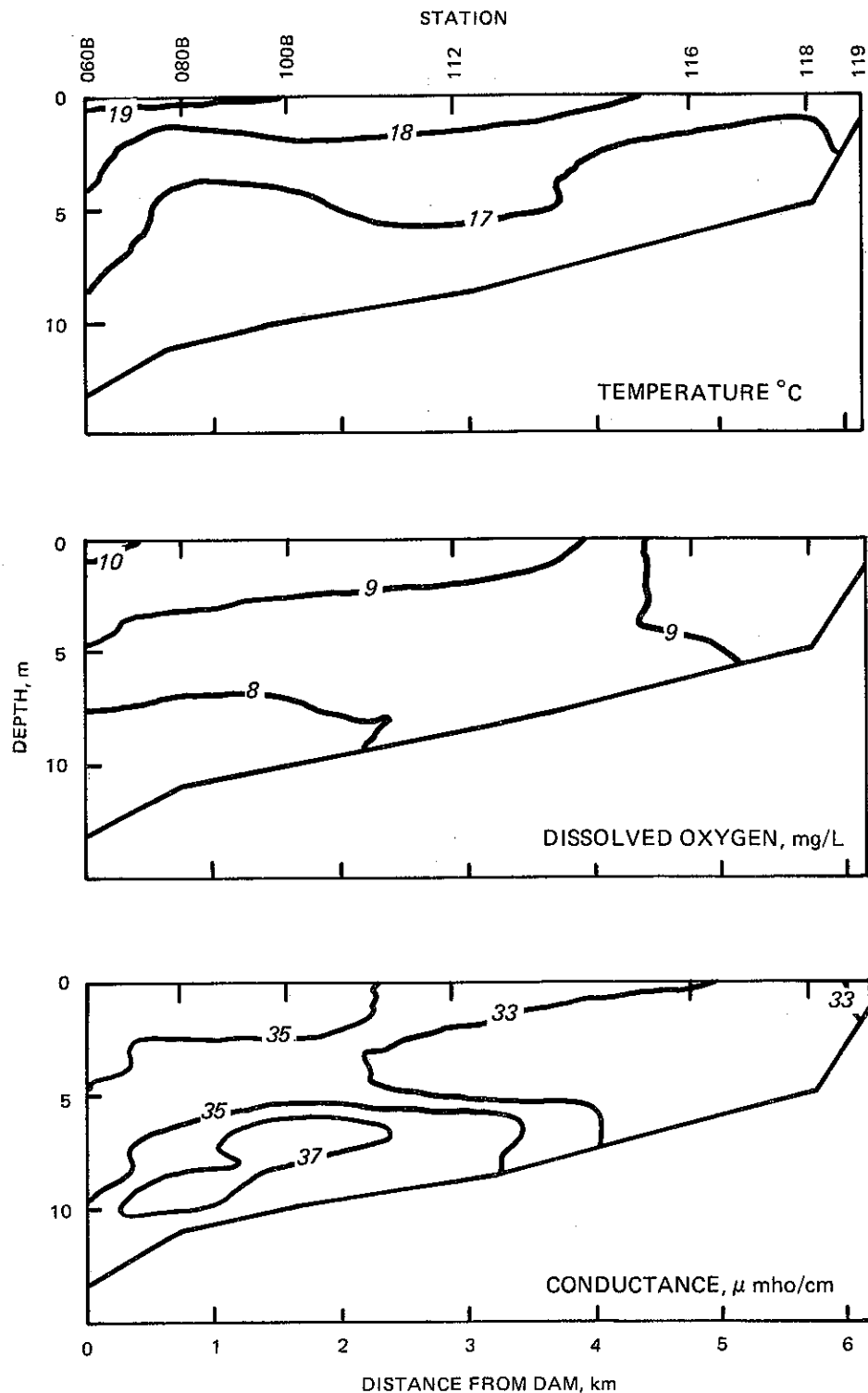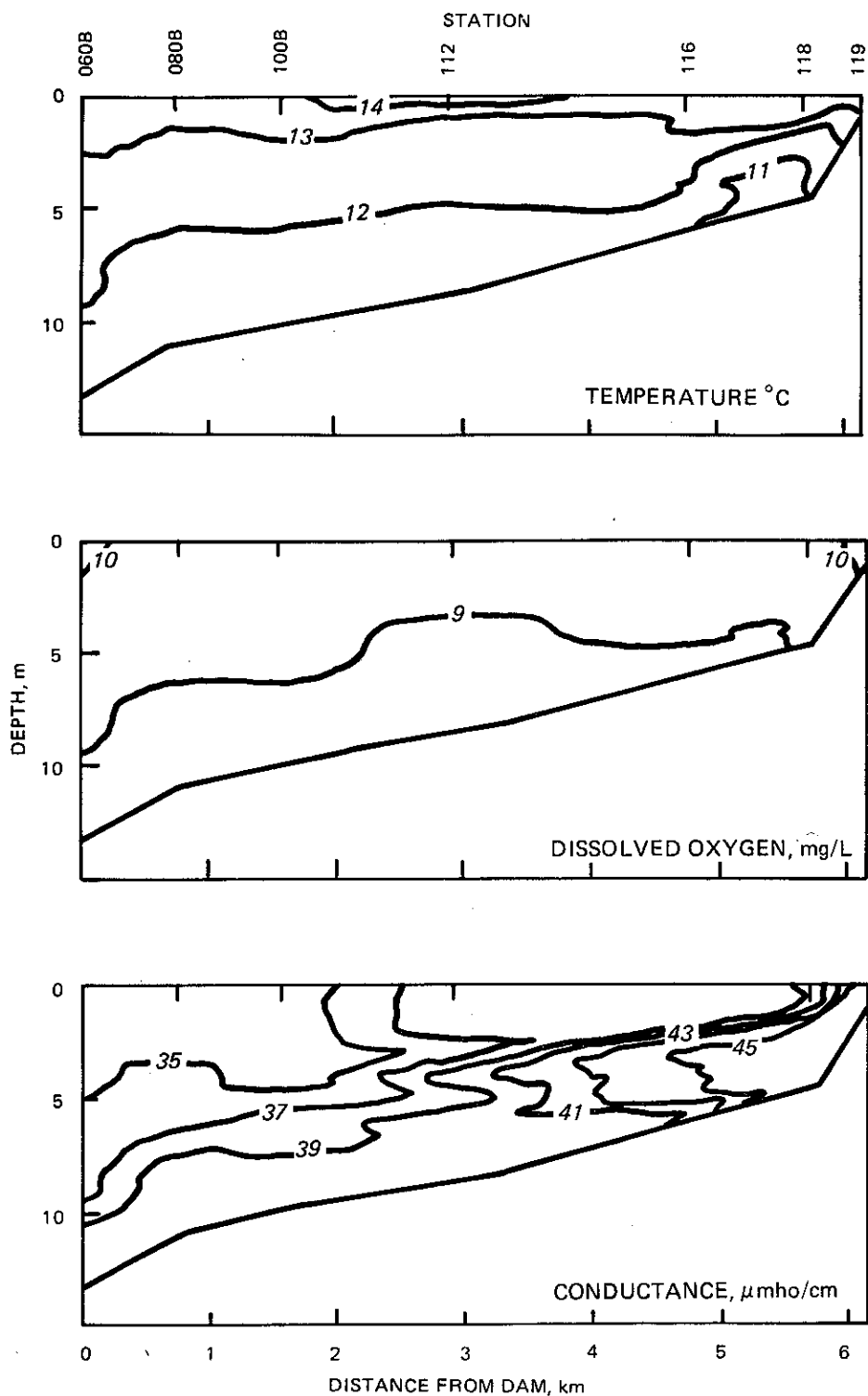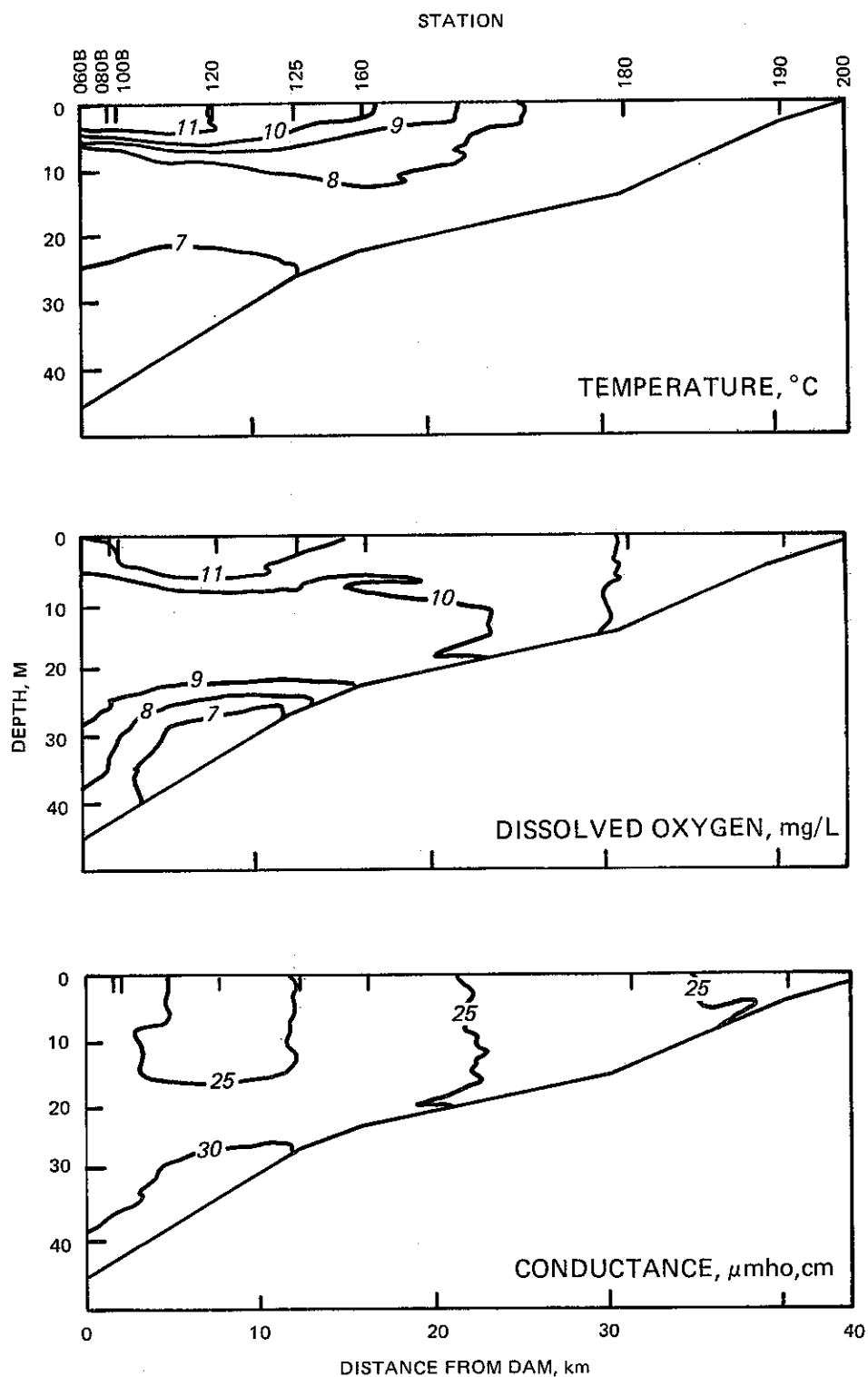Lake on 22 November 1983

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STATION

Figure 9. Vertical and longitudinal patterns in tempera-
ture (upper panel), dissolved oxygen (middle panel), and
specific conductance (lower panel) in Richard B. Russell
Lake on 22 February 1984

Also apparent during this phase were higher concentrations of nutrients and metals in the embayments and at Station 120. These occurrences suggest the significance of loadings via secondary tributaries. As a result of these loads and the inundation of terrestrial vegetation and soils, the depletion of dissolved oxygen stores in bottom waters was observed intermittently at Station 120 and at other sites throughout each of the embayments. As will be discussed later, these conditions became well established with continued increases in pool elevation and stratification.

Pool elevation increases during Phase III inundated approximately 7,650 ha of bottomlands containing large quantities of organic material and readily solubilized nutrients and metal. While it was anticipated that significant chemical changes would result from the inundation of soils and vegetation, only moderate concentration increases were observed in the water column. For example, total organic carbon concentrations at Station 60 increased from a mean of 1.4 mg/l on 13 October 1983, to 4.1 mg/l on 15 December 1983. Concentrations then declined to 2.0 mg/l on 8 February 1984. Changes in the concentrations of total and dissolved nitrogen, total phosphorus, and total iron displayed similar temporal patterns. However, total and dissolved manganese, dissolved phosphorus, and dissolved iron remained low in concentration. As will be discussed later, the small magnitude of these increases may have been related, in part, to a lack of thermal stratification, dissolved oxygen conditions, and a rapid hydrological flushing.

Limnological conditions during the stratified period. Patterns of thermal stratification in Richard B. Russell Lake were influenced by meteorological events, basin morphometry, and hydrological events. During the stratified period, pool elevation was maintained at approximately 140 m MSL and releases were discharged through near-surface tainter gates from May until December. Because of basin morphometry and hydrologic conditions, spatial and temporal variations in the development of thermal stratification were observed. In general, stratified conditions developed most rapidly in upstream areas and embayments where basin morphometry was shallow. Lakewide surface heating was apparent in late March, and stratified conditions were well established throughout the lake by June. However, epilimnetic depth was shallow (3-5 m) throughout the stratified period due to the influence of near-surface tainter gate releases and the lower pool elevation. In addition, thermal stratification was disrupted in the upper reaches of the main basin due to

cool discharges from Hartwell Lake during power generation.

Warming trends in February and March resulted in the early establishment of stratified conditions in the Rocky River and Beaverdam Creek embayments. This was observed from in-situ data collected on 22 March 1984 at several stations located along the length of each embayment and at their confluence with the main basin (i.e., Station 120) (Figure 10). Stratified conditions were evident from the headwaters of each embayment to near the confluence on this date, with an epilimnion established to a depth of 3-5 m. This was due, in part, to a narrow channel morphology in the embayments and the maintenance of a pool elevation 1.5 m below normal pool. As a result, surface areas were small in the embayments and more protected from wind-induced mixing. Locations near the confluence had greater surface areas and were thus more susceptible to mixing.

Continued warming in March and April resulted in temperature increases in the Rocky River and Beaverdam Creek embayments and the occurrence of stratification by May. On 14 May 1984, surface temperatures ranged from 23-25 deg. C along the length of each embayment and at Station 120. Bottom temperatures were lowest at Station 120 (11 deg. C), while upstream embayments exhibited bottom temperatures near 17 deg. C.

In June, thermal stratification was well established in the two embayments. Data collected on 11 June indicated marked stratification patterns from the headwater regions of Rocky River and Beaverdam Creek to the confluence at Station 120 (Figure 10). Surface temperatures were approximately 28-30 deg. C while bottom temperatures ranged from near 22 deg. C at the head of each embayment to 12 deg. C at Station 120. The epilimnion was only 2-5 m thick during this period, and there were marked temperature gradients in the metalimnion. This pattern was noted throughout the stratified period. On 11 June, temperatures changed from 19.6 deg. C to 15.0 deg. C within a 2-m depth range at Station 120.

Similar patterns in the development of thermal stratification were observed along the longitudinal axis of the main basin. Surface heating and intermittent mixing resulted in weakly stratified conditions during late February to mid-March. On 19 March 1984, surface temperatures were longitudinally uniform from Station 60 to Station 160 (14-15 deg. C). Epilimnetic thickness was shallow (1 to 2 m), and bottom temperatures ranged from 7 to 9 deg. C (Figure 11).
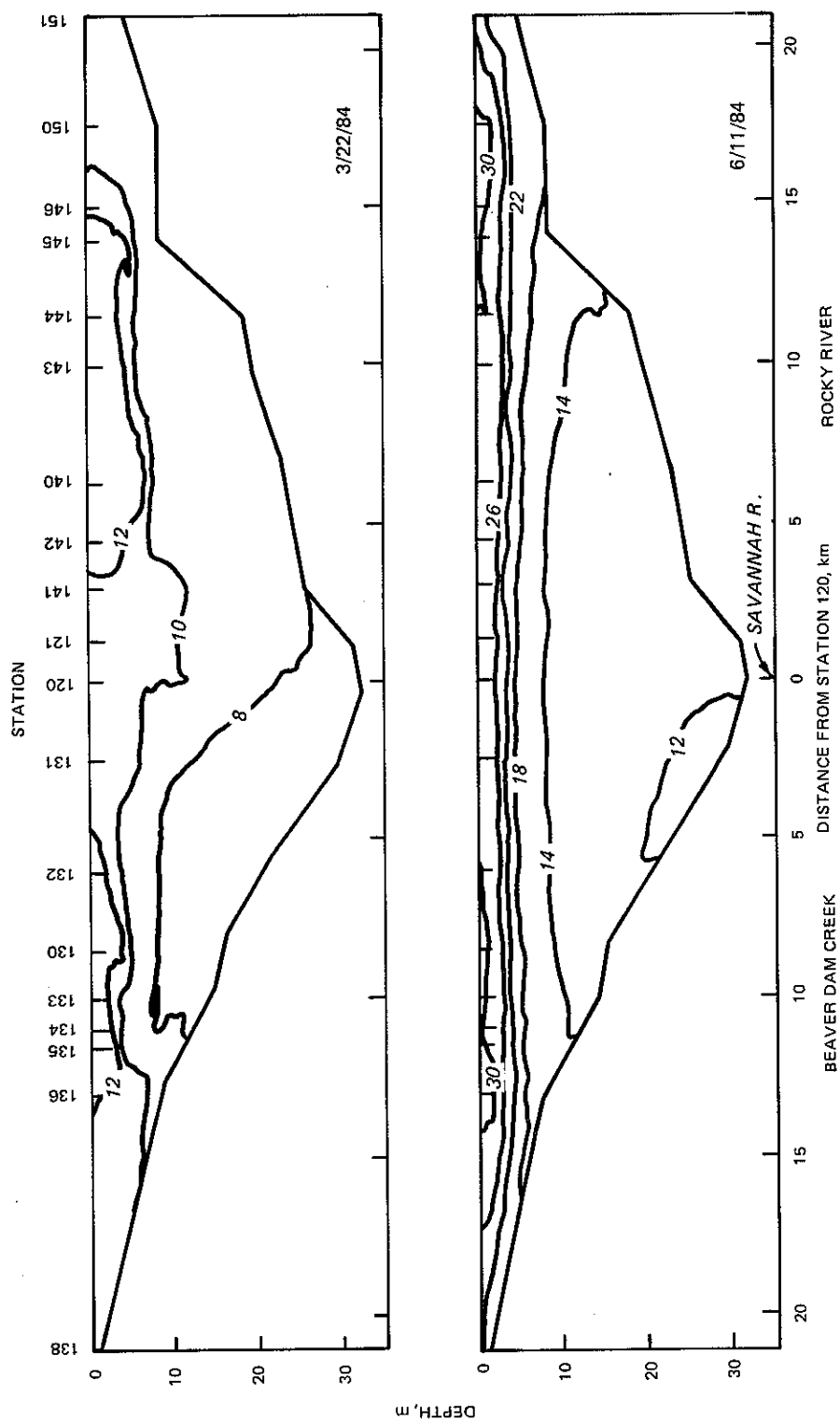
Figure 10. Vertical and longitudinal patterns in temperature (deg. C) for Beaverdam Creek and Rocky River embayments

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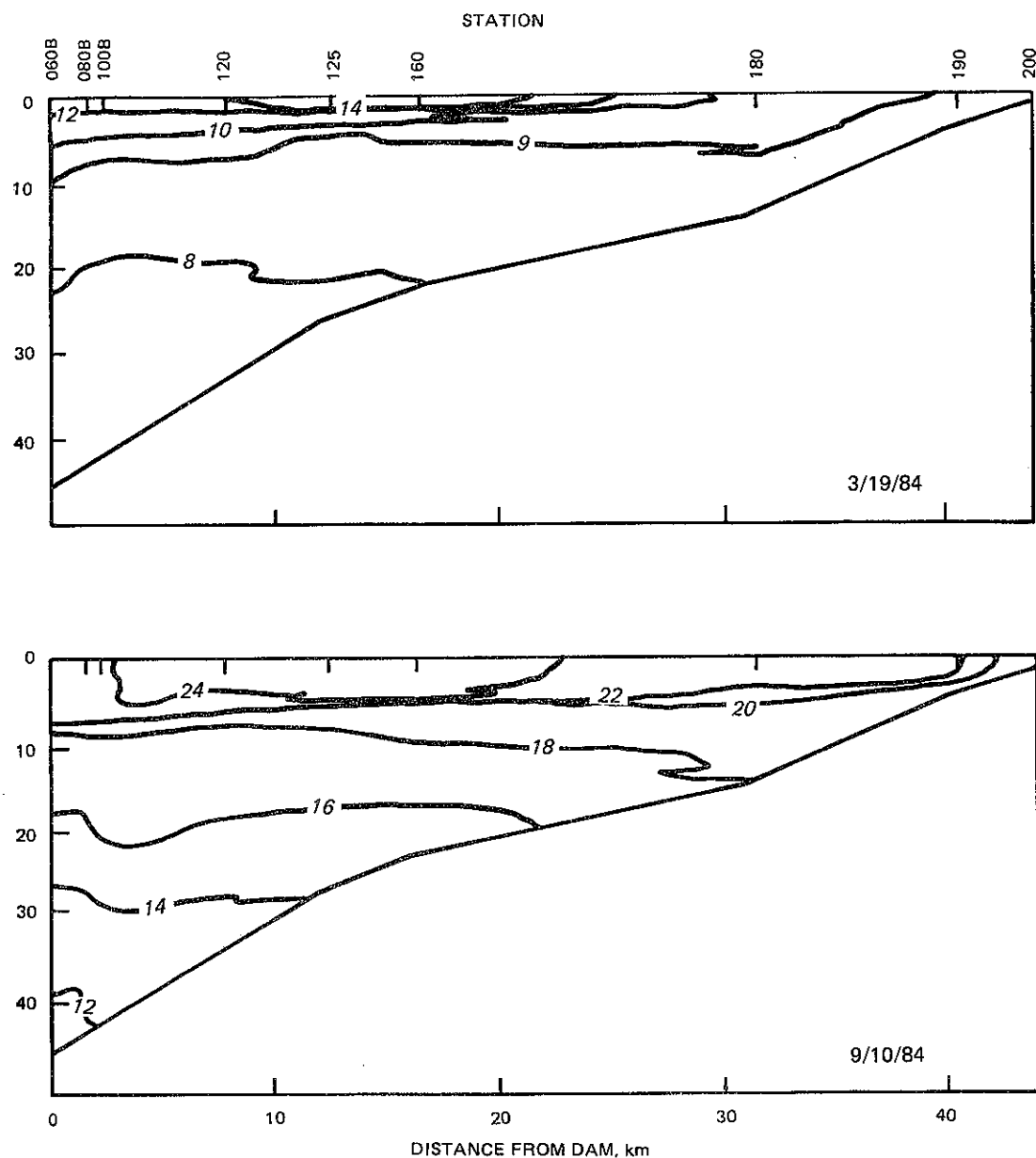
Figure 11. Vertical and longitudinal patterns
in temperature (deg. C) for the main basin of
Richard B. Russell Lake

Surface warming resulted in the development of marked thermal stratifi-
cation along much of the main basin from Station 60 to near Station 180. Data
collected on 10 September were representative of the longitudinal and vertical
patterns in thermal structure observed throughout the summer stratification
period. Surface waters had reached a maximum temperature of approximately
30 deg. C, and the epilimnetic depth was established at 3-5 m in June and

July. By September, surface temperatures had decreased to 22-24 deg. C, but stratification was still evident (Figure 11).

Discharges from Hartwell Lake had a pronounced influence on thermal patterns above Station 160 during the stratified period. The cooler temperatures of these middepth releases greatly modified temperatures at Stations 190 and 200, and generated an interflowing density current in the vicinity of Station 180. As will be discussed later in Part V, highly dynamic interactions occur between Hartwell Dam releases and the receiving waters on a diel cycle.

Apparent in the main basin, and in the Rocky River and Beaverdam Creek embayments, was the occurrence of a shallow (3- to 5-m-thick) epilimnion and marked temperature gradients in the metalimnion throughout the stratified period. Near-surface tainter gate releases and a lower lake level appeared to regulate thermal structure. Tainter gate releases were employed throughout the summer stratified period to improve water quality conditions of the discharges. However, the location of the withdrawal zone was only 5 to 20 m below the lake's surface due to the maintenance of a pool elevation 1-2 m below normal pool. In-pool flow patterns created by tainter gate releases resulted in the removal of epilimnetic water within a shallow withdrawal zone and replacement with cooler inflowing waters originating from Hartwell Lake and the embayments. These flow events limited epilimnetic expansion and prevented hypolimnetic flushing.

Associated with the development of thermal stratification was the lake-wide occurrence of dissolved oxygen depletion in the hypolimnion. The extent to which dissolved oxygen was depleted was related to (1) basin morphometry, (2) patterns of thermal stratification, (3) water residence time, (4) the presence of large quantities of oxidizable organic material, and (5) material loads from tributaries. Dissolved oxygen depletion developed rapidly in the shallow embayments and spread to the confluence of the two tributaries, and then toward the dam and Station 160.

Dissolved oxygen depletion began in the two major embayments early in the stratified period. Anoxic or near-anoxic conditions were first observed at Beaverdam Creek and Rocky River stations on 22 March, shortly after the onset of thermal stratification (Figure 12). The most severe depletion occurred in the upper reaches of each embayment (i.e., near Stations 130 and 140) and progressed toward Station 120. This pattern was related, in part, to the shallowness of these basins, the influence of organic material loading from

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Figure 12. Vertical and longitudinal patterns in dissolved oxygen (mg $O_2/1$) for Beaverdam Creek and Rocky River embayments

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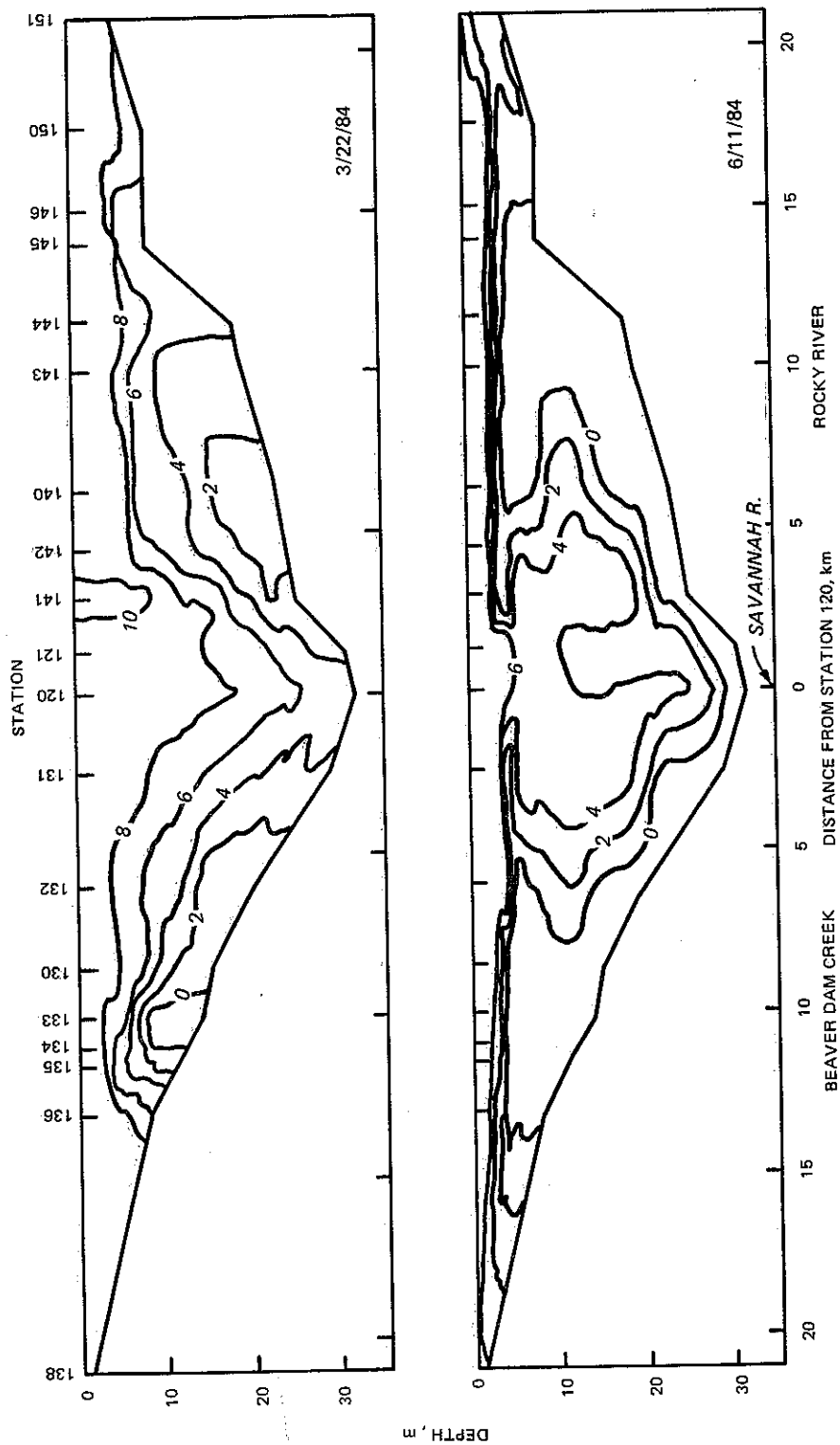the inflows, and the decomposition of large quantities of organic materials in inundated soils and vegetation.

In June, the two tributary embayments exhibited anoxia in bottom waters along much of their length (Figure 12). In the shallow areas of each embayment, anoxic or near-anoxic conditions were evident from the sediment surface to the bottom of the thermocline. Marked dissolved oxygen depletion was also evident below the thermocline at Station 120. Also apparent was the establishment of distinct metalimnetic dissolved oxygen minima at downstream locations of each embayment.

In the main basin, dissolved oxygen depletion was first observed at the confluence of the two tributary embayments (Station 120) and in the forebay area (Station 60) (Figure 13). Patterns in the distribution of dissolved oxygen on 19 March 1984 reflected the impact of a significant demand on hypolimnetic oxygen stores at Station 120. Concentrations in the bottom waters on this date were less than 4.0 mg/l. By September 10, a large zone of anoxia was established from the dam to Station 160, and anoxic conditions were evident from bottom to near the thermocline. Concentrations in the epilimnion on this date ranged from 7 to 8 mg/l.

Hartwell discharges had a persistent influence on dissolved oxygen concentrations in the upper end of Richard B. Russell Lake during the stratified period. In early summer, hypolimnetic discharges from Hartwell Dam were high in dissolved oxygen. However, later in the summer, dissolved oxygen depletion in Hartwell Lake resulted in the release of water low in dissolved oxygen. When releases were minimal, dissolved oxygen concentrations were high in the headwater region of Russell Lake. However, dissolved oxygen concentrations often decreased to near 5 mg/l from Station 180 to Hartwell Dam during power generation.

Richard B. Russell Lake pH patterns exhibited minimal spatial or temporal fluctuations. At Station 60, pH ranged from 6.5 to 7.4 at the surface and from 6.1 to 6.9 at the bottom depth. Similar ranges in pH values were observed at other stations.

Marked temporal, vertical, and spatial patterns in chemical conditions were documented during the first year of impoundment. These patterns were the result of the occurrence of a variety of chemical and biological processes. Concentrations were also influenced by thermal structure and hydrodynamics. Since inundated soils and vegetation contained a large pool of nutrients and
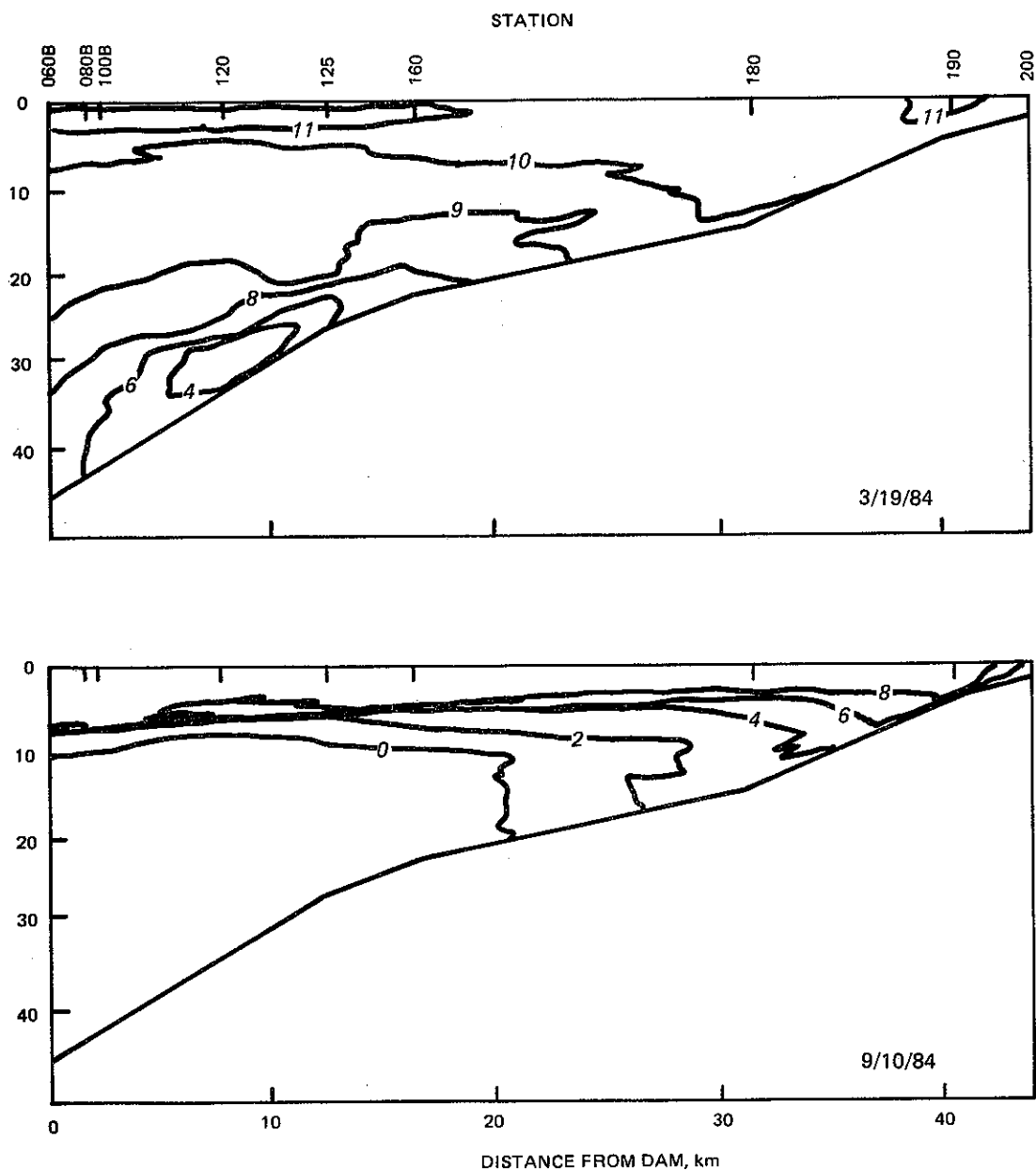
Figure 13. Vertical and longitudinal patterns in
dissolved oxygen (mg O_2/l) for the main basin of
Richard B. Russell Lake

metals, interactions between soils and bottom waters led to the establishment
of many of the chemical patterns observed in the hypolimnion. Chemical con-
centrations were also affected by anoxic conditions in bottom waters and a
rducing environment.

Seasonal and spatial trends in oxidation-reduction potential were
associated with dissolved oxygen patterns in the hypolimnion. Negative

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oxidation-reduction potentials were first detected on 2 April in the bottom waters of the two embayments (i.e., Station 130 and 140), where hypolimnetic dissolved oxygen depletion occurred rapidly. The occurrence of anoxia in the forebay area and at Station 120 led to the establishment of a reducing environment in these hypolimnetic areas by late May. In June, negative potentials were observed in deep portions of the main basin and embayments. Negative potentials were also observed at Stations 150 and 160 during June. However, intermittent shifts in the potential occurred throughout the stratified period due to the influences of density flows and inflows during storm events.

Vertical and longitudinal patterns of redox potential were evident on 30 July (Figure 14). On this date, negative potentials were observed at bottom depths from Station 60 to Station 160. Station 120 exhibited the largest zone of negative potentials in the hypolimnion, which may have been related to its proximity to the two major embayments. Above the zone of dissolved oxygen depletion and upstream of Station 160, redox potential was always positive. In general, these spatial and vertical patterns developed throughout the
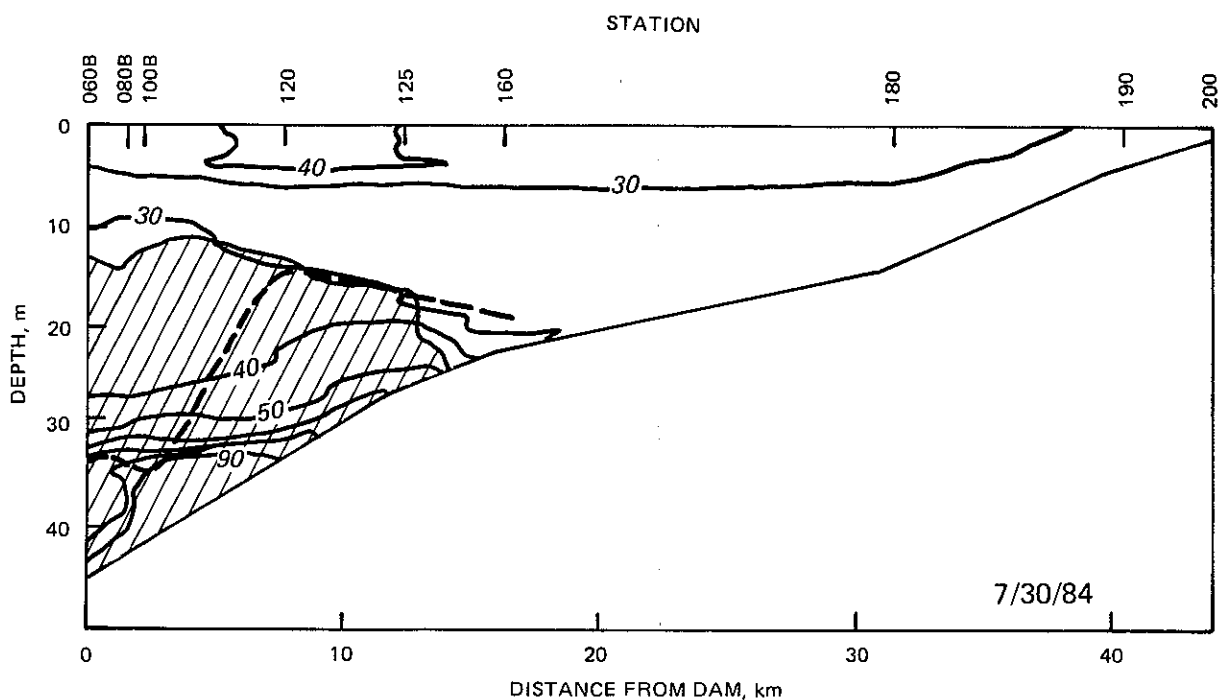

Figure 14. Vertical and longitudinal patterns in specific conductance (umhos/cm) for the main basin of Richard B. Russell Lake on 30 July 1984. Shaded area represents dissolved oxygen concentrations less than 1.0 mg/l. Dashed line indicates depth where redox potential changes from a positive to a negative value

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stratified period as hypolimnetic areas became depleted in dissolved oxygen.

Specific conductance values, a gross indicator of dissolved material concentrations, reflected the development of thermal and chemical stratification within the reservoir. Specific conductance values increased in the bottom waters of the two embayment stations shortly after the establishment of stratified conditions and anoxia. However, despite the occurrence of anoxia in the extreme upstream end of each embayment during spring, bottom water values were low, reflecting the influence of inflows from secondary tributary streams. Specific conductance increased in the forebay area and at Station 120 beginning in May and June. Bottom values on 25 June ranged from 60 umhos/cm to 71 umhos/cm at Stations 60 and 120, respectively; values at Stations 130 and 140 ranged from 79 umhos/cm to 69 umhos/cm, respectively. By July, high specific conductance values (i.e., >50 umhos/cm) were detected in the hypolimnion from the dam to Station 160, and in a major portion of the two embayment areas.

Data collected on 30 July were characteristic of the longitudinal and vertical patterns observed in the main basin (Figure 14). Values ranged from over 90 umhos/cm to near 35 umhos/cm at the bottom depths of Station 60 and 160, respectively. The vertical extent of specific conductance increases was suggested by the 30 and 35 umhos/cm contour lines in Figure 14. Epilimnetic values were relatively low and uniform, while hypolimnetic values increased rapidly with increases in depth.

Spatial patterns in specific conductance values indicate the possible occurrence of interflows at the depth of the tainter gate withdrawal zone. For example, specific conductance exhibited a zone of uniform values on 30 July which extended from the metalimnetic area of the forebay region to the headwaters. Values within this zone were lower than values observed in either the epilimnion or the hypolimnion, but comparable to levels observed at Station 200. Within this depth interval, specific conductance ranged from 28 to 35 umhos/cm in the main basin while Station 200 exhibited a value of 26 umhos/cm. This observation indicates that Hartwell water may have been moving through Russell Lake as an interflow confined to depths near the depths of the tainter gate withdrawal zone. This contention is supported by similar observations on other dates during the stratified period. Also, a number of chemical variables exhibited a distinct band of uniform concentrations within the metalimnion.

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Organic carbon concentrations exhibited marked spatial patterns during the stratified period. These patterns were related to algal productivity, the quality of release waters from Hartwell Dam, and interactions between bottom waters and inundated soils. Surface waters exhibited peaks in total and dissolved organic carbon concentration in the headwater region in June and July (Figure 15). Concentration increases were also observed in the vicinity of Stations 120 and 60. Peaks at upstream locations were related to Hartwell Lake releases while increases in the lower portion of the pool coincided with increases in chlorophyll a concentrations.

Organic carbon concentrations also displayed pronounced seasonal and longitudinal patterns in bottom strata during the stratified period. In the main basin, total organic carbon concentrations began to increase in the forebay area shortly after the onset of thermal stratification (Figure 15). Elevated levels of total organic carbon were apparent from Station 60 to Station 160 by late May. However, highest concentrations were routinely observed in the forebay area, with gradients of decreasing values toward Station 160. Dissolved organic carbon exhibited similar spatial trends during the stratified period.

Distinct vertical and longitudinal patterns in total organic carbon and dissolved organic carbon concentrations were observed in the main basin on 10 September 1984 (Figure 16). On this date, total and dissolved organic carbon concentrations were highest at the bottom depths, and pronounced vertical gradients were apparent in the deep downstream portion of the lake. Few longitudinal differences were observed. An exception occurred in the upper strata where increased algal biomass, as evidenced by chlorophyll a concentrations, led to elevated organic carbon concentrations. Also apparent was the fact that much of the organic carbon in the lake was in a dissolved form.

Interflowing density currents and release patterns at Hartwell Dam had an influence on organic carbon concentrations near the metalimnetic area (Figure 16). Total and dissolved organic carbon concentrations here were low and comparable to concentrations of discharge waters from Hartwell Dam. This suggests that inflows from Hartwell Lake move through Russell Lake at the depth of the thermocline.

Organic carbon concentrations were also elevated at the two embayment stations during the stratified period. At the bottom depths at Stations 130 and 140, total and dissolved organic carbon concentrations rapidly increased

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Figure 15. Temporal and longitudinal patterns in total and dissolved organic carbon (mg C/l) at the 1-m depth (upper panels and at the bottom depth (lower panels) for the main basin of Richard B. Russell Lake

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Figure 16. Vertical and longitudinal patterns in total and
dissolved organic carbon for the main basin of Richard B.
Russell Lake on 10 September 1984

after the onset of thermal stratification (Figure 17). Total organic carbon
concentrations increased from April to June reaching maxima of 16.2 mg C/l
and 10.4 mg C/l at Stations 130 and 140, respectively. Total and dissolved
organic carbon then declined from May until September.

Spatial differences were apparent between the embayment stations and
the main basin during the stratified period. In general, epilimnetic and hy-
polimnetic concentrations were higher in the two embayments than in the main
basin, and concentrations increased more rapidly in the bottom strata at the
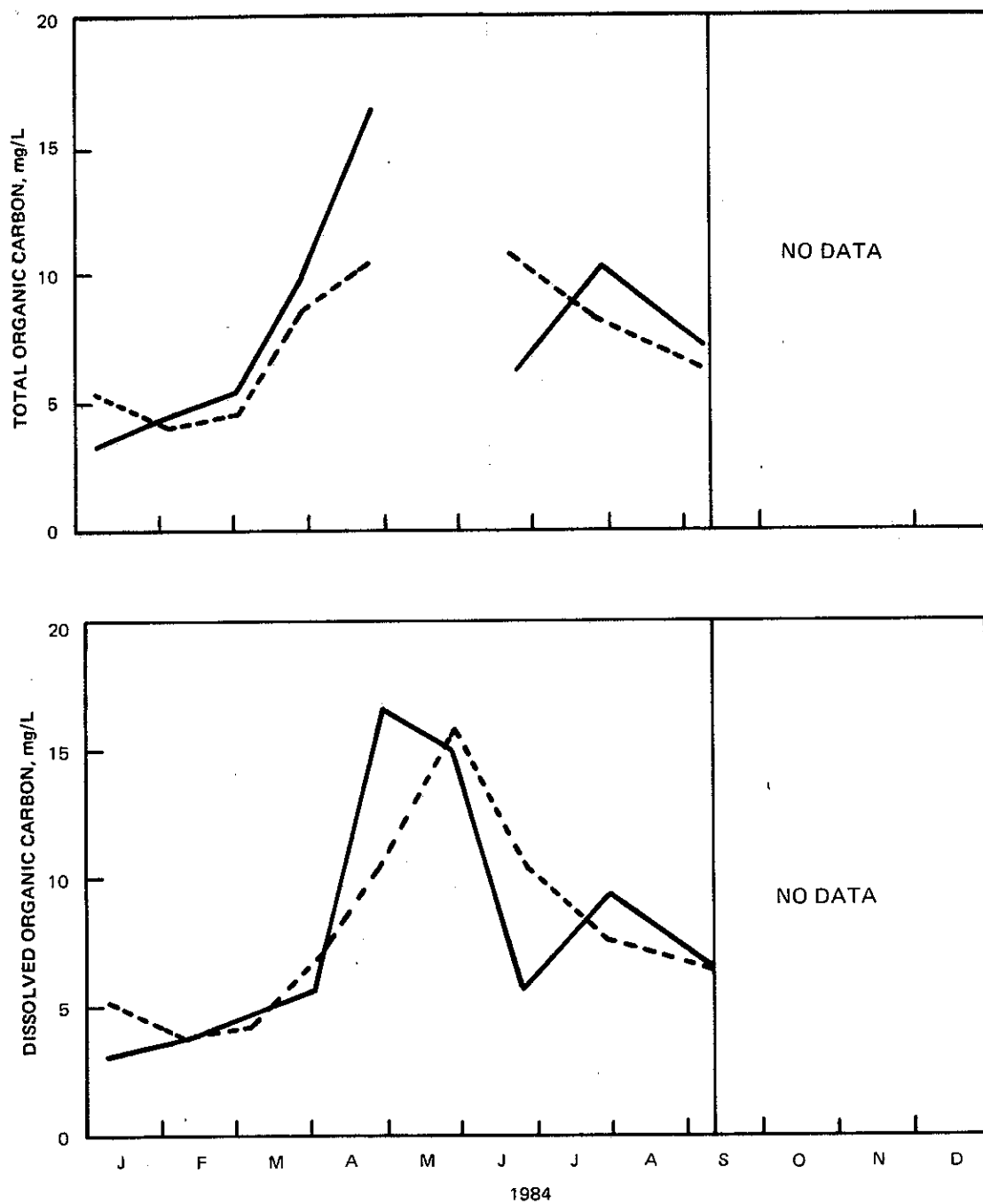
Figure 17. Temporal changes in total and dissolved
organic carbon at the bottom depth of Stations 130
(solid line) and 140 (broken line)

two embayment stations. These observations were related to localized differences in organic matter inputs to the epilimnion and hypolimnion, and to a higher sediment area:hypolimnetic volume ratio in the two embayments. These differences would tend to enhance sediment/water interactions at the shallow embayment stations, resulting in a higher and more rapid accumulation of organic carbon in the hypolimnion.

Total nitrogen, the sum of organic nitrogen, ammonia, nitrate, and nitrite nitrogen, exhibited patterns similar to those for organic carbon (Figure 18). Total nitrogen concentrations increased slightly during the initial filling period, due to the inundation of organic material, and declined until the onset of thermal stratification. During the stratified period, surface concentrations increased to a seasonal maximum in the headwater region in early June due to inflows, and smaller surface concentration increases were observed at midreservoir from May until July and in September. Total nitrogen concentrations increased at the bottom depth shortly after the occurrence of thermal stratification. Concentration increases were marked in the forebay area from May until September, and longitudinal gradients were evident from the dam to Station 160 by August.

Total dissolved nitrogen and ammonia-nitrogen exhibited similar seasonal and longitudinal patterns. These soluble forms also comprised a large percentage of the total nitrogen accumulating at the bottom depths. Ammonia-nitrogen, a chemical end product of microbial reduction of nitrate and nitrite, and of the decomposition of organic material, was observed in substantial amounts in the hypolimnion during the stratified period (Figure 19). Ammonia-nitrogen concentrations began increasing at the bottom depths after stratification and the establishment of hypolimnetic anoxia in May. By September, the lower half of Richard B. Russell Lake experienced elevated concentrations ranging from 1.03 mg N/1 to 0.44 mg N/1 at Stations 60 and 160, respectively. Surface concentrations of ammonia-nitrogen also exhibited peaks in the lower half of Richard B. Russell Lake during the filling period and during the onset of thermal stratification; thereafter, levels were generally low. A small peak was also detected in the Hartwell Dam tailwater region in late May.

Vertical and longitudinal patterns in nitrogen distribution further suggest that inundated bottomlands were an important source of nitrogen to the hypolimnion during the stratified period. A large zone of accumulated
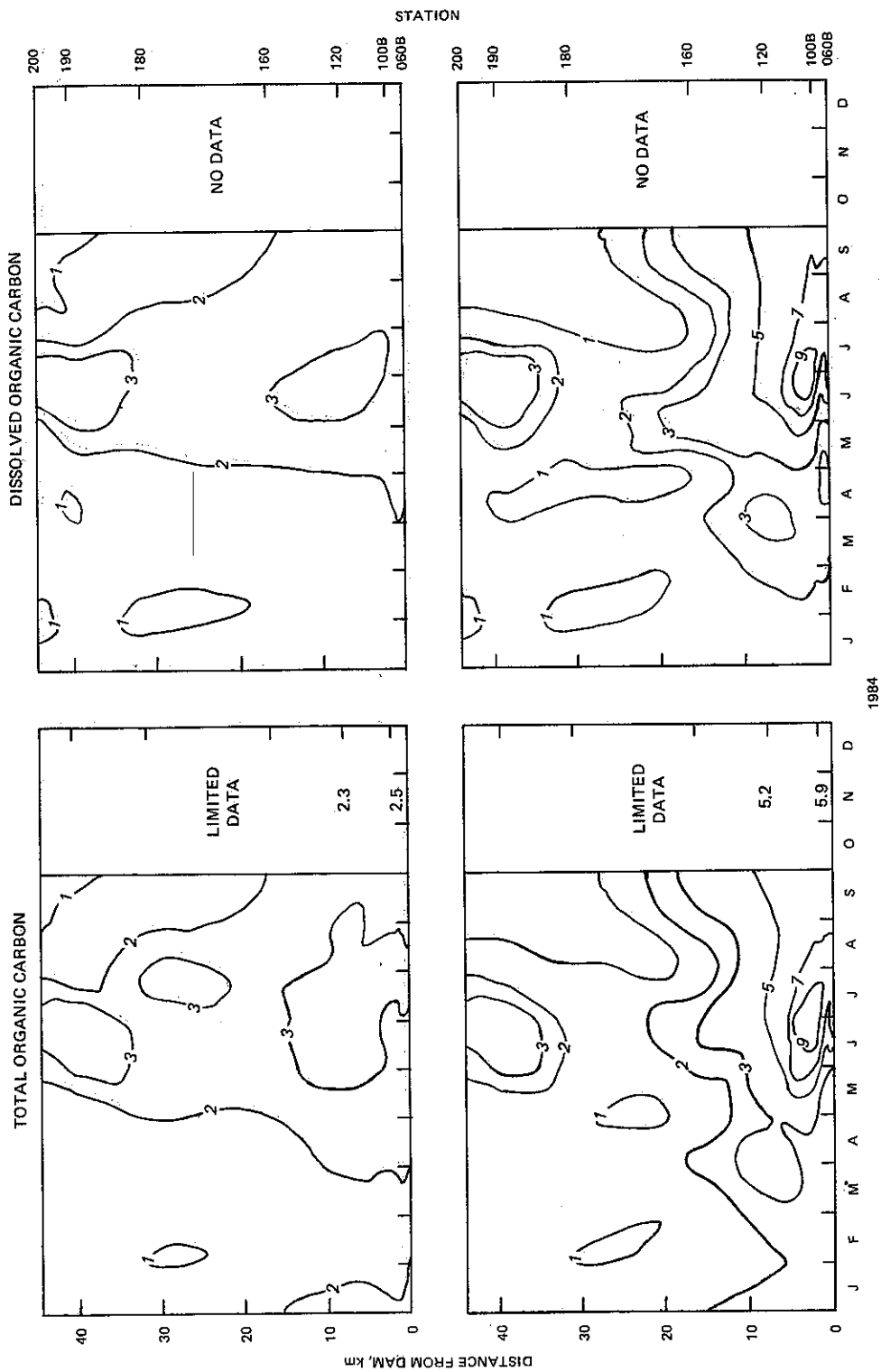
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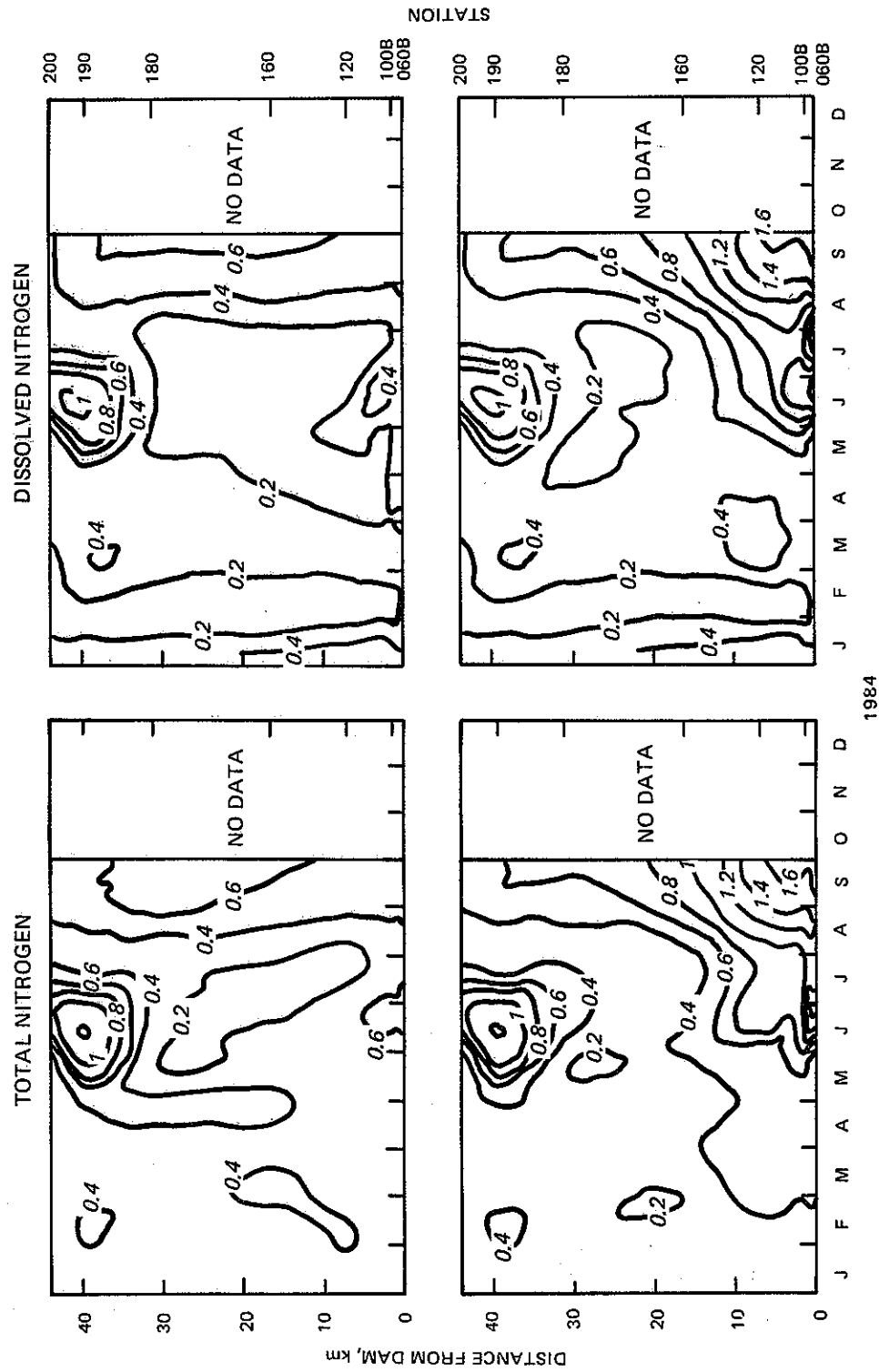
Figure 18. Temporal and longitudinal patterns in nitrogen forms (mg N/1) at the 1-m depth (upper panels) and at the bottom depth (lower panels) for the main basin of Richard B. Russell Lake
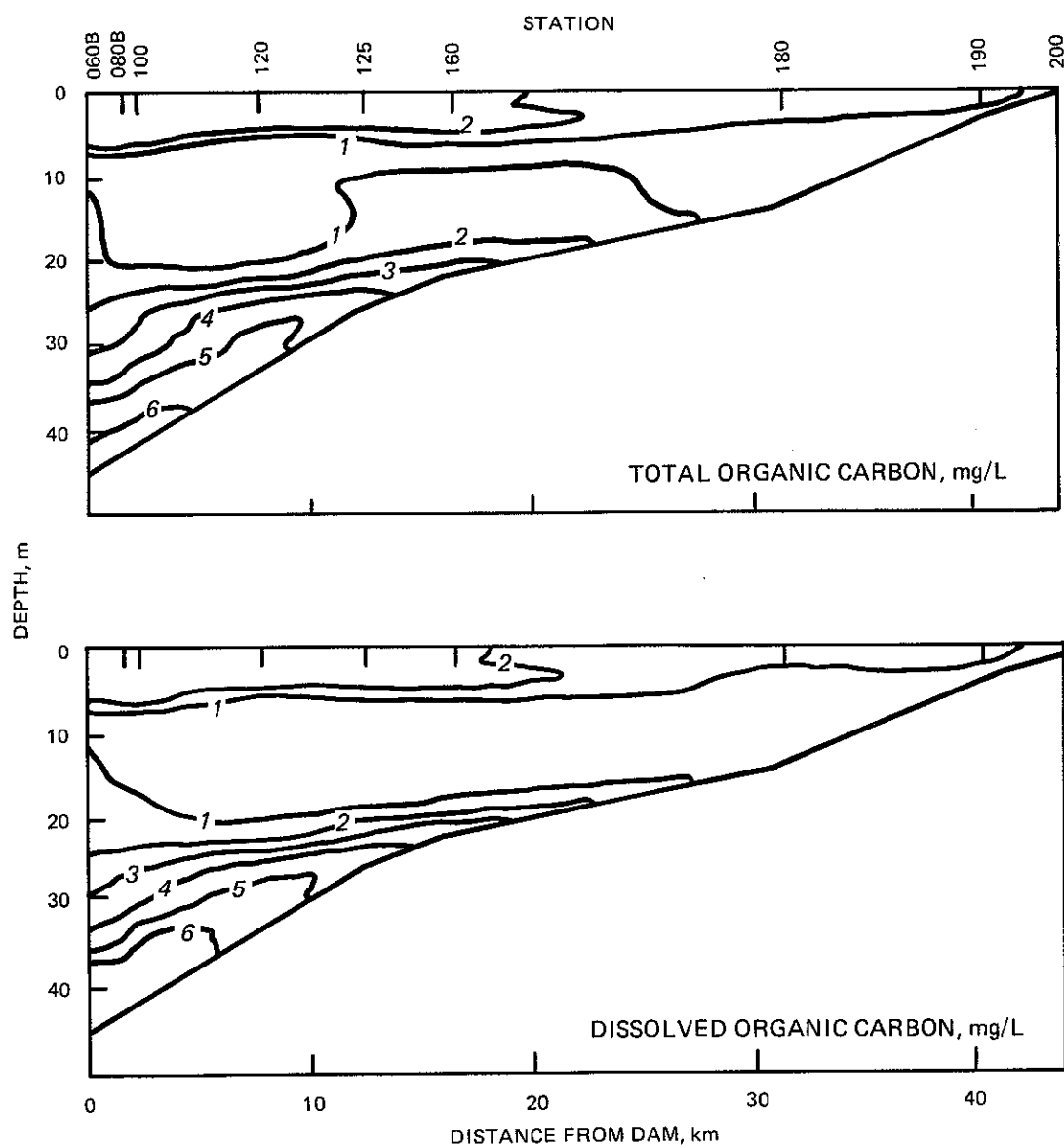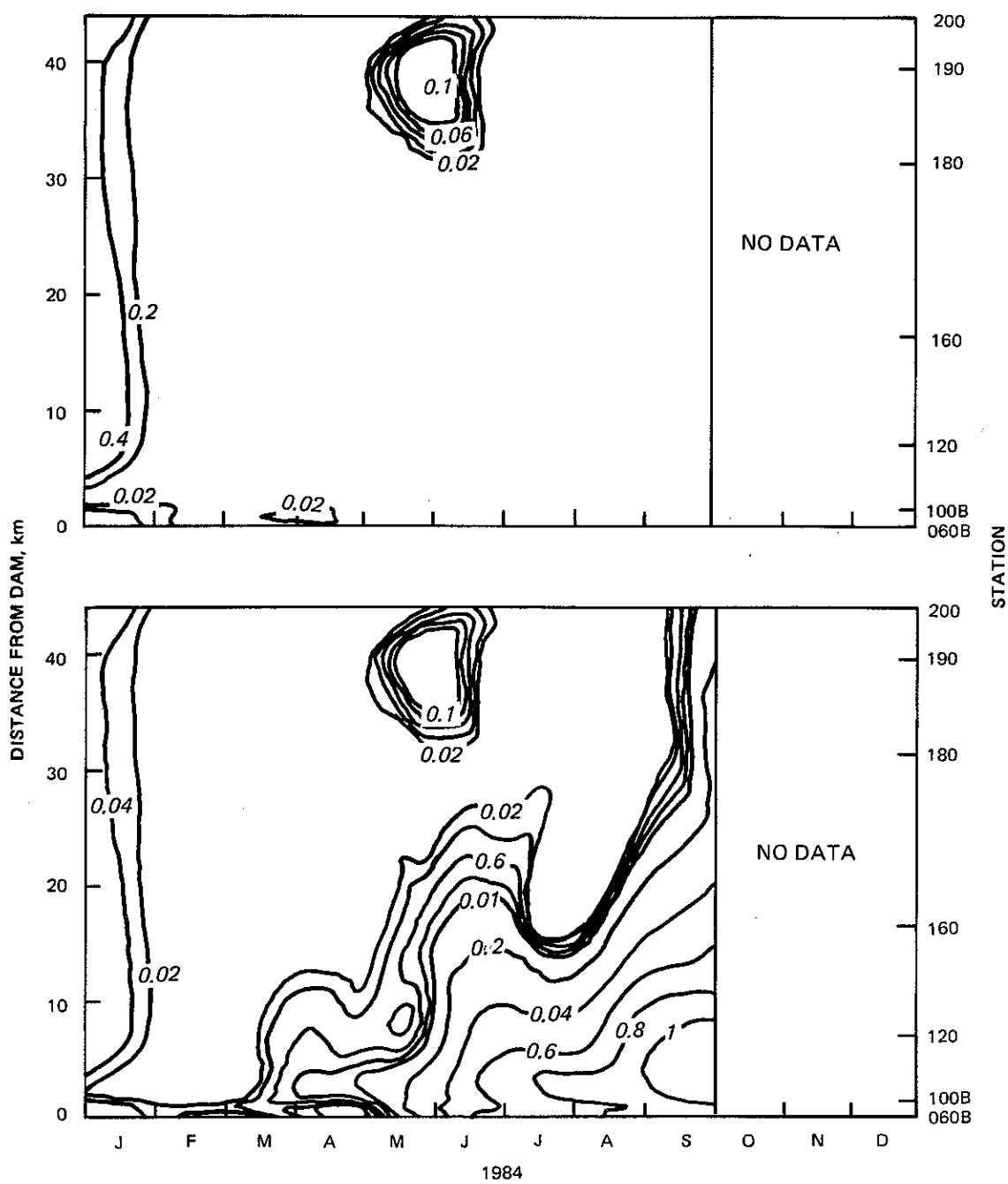
Figure 19. Temporal and longitudinal patterns in nitrate-plus-nitrite nitrogen (upper panel) and ammonia-nitrogen (lower panel) at the bottom depth for the main basin of Richard B. Russell Lake. All values expressed as mg N/l

nitrogen was evident in the hypolimnion from Station 60 to Station 160 by
10 September 1984 (Figure 20). Concentrations were highest near the sedi-
ment surface, and vertical gradients were apparent. High concentrations of
dissolved nitrogen and ammonia-nitrogen also indicated that a large percent-
age of the hypolimnetic nitrogen was in a soluble form (Figures 20 and 21).

Nitrate and nitrite nitrogen is the product of bacterial oxidation
and generally occurs under aerobic conditions. This form, measured as
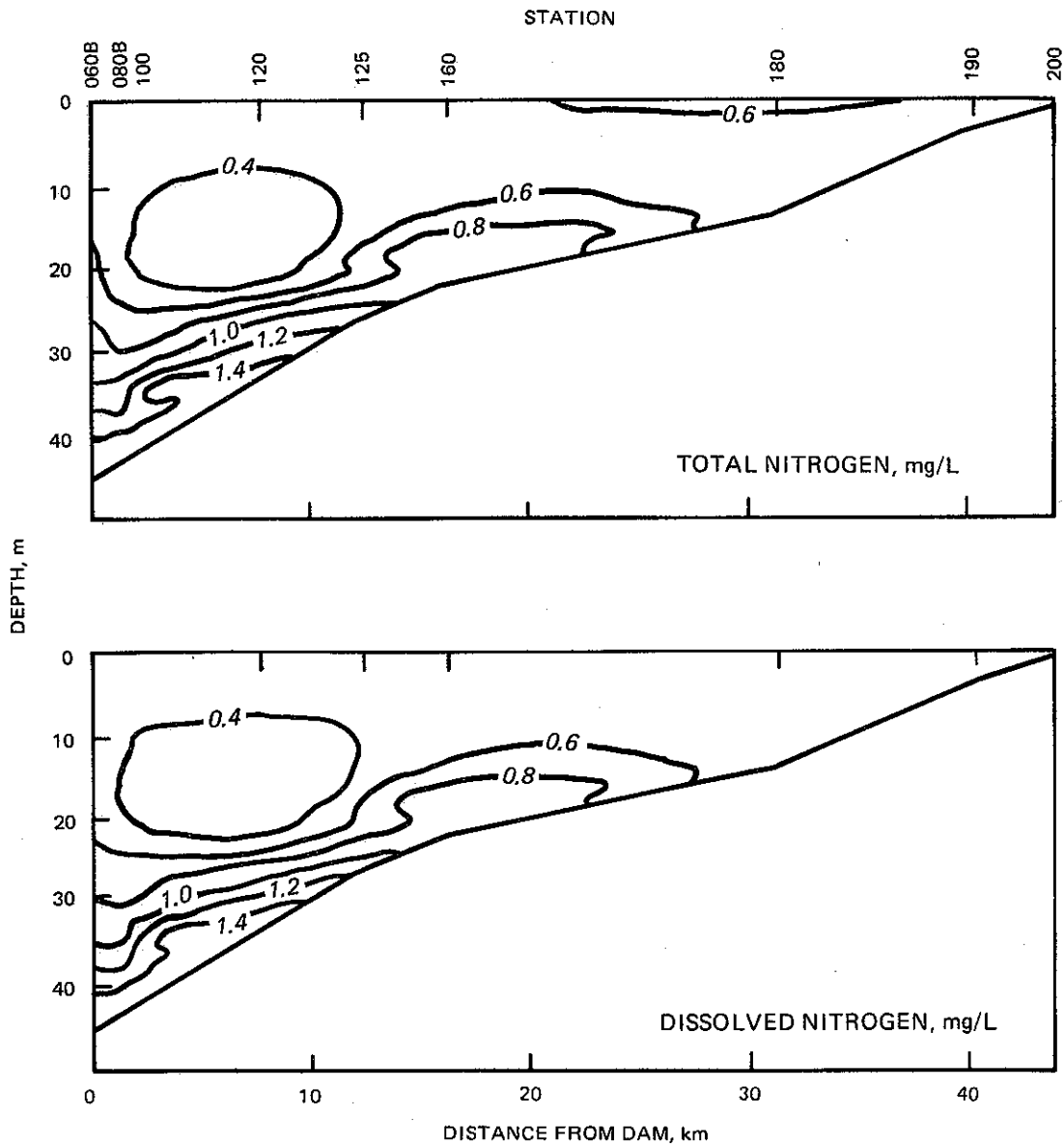
Figure 20. Vertical and longitudinal patterns in total
and dissolved nitrogen for the main basin of Richard B.
Russell Lake on 10 September 1984
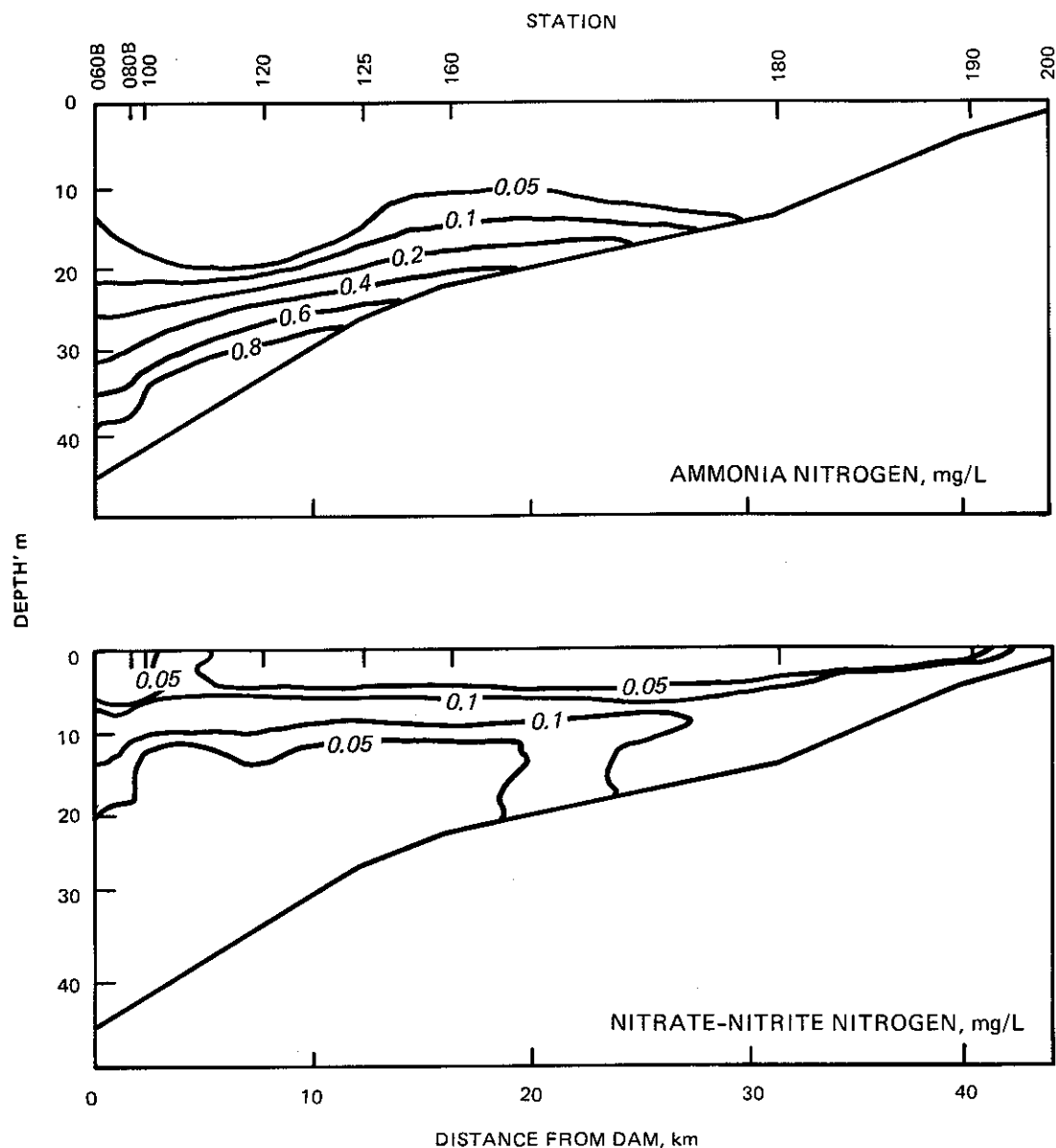
Figure 21. Vertical and longitudinal patterns in ammonia-
nitrogen and nitrate-nitrite nitrogen for the main basin
of Richard B. Russell Lake on 10 September 1984

nitrate-plus-nitrite nitrogen, displayed seasonally low concentrations in the
surface and bottom waters. Since nitrate is an important requirement for
algal growth, summer declines in the surface waters were probably the result
of uptake and assimilation by algae. Surface concentrations were undetectable
from May until September in much of the main basin. A concentration increase
was detected in the headwater region in early June, possibly due to the in-
fluence of releases from Hartwell Dam. Seasonal decreases in the bottom

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waters usually follow bacterial reduction of nitrate to ammonia under anaerobic conditions. Therefore, concentrations at this depth interval were undetectable in areas experiencing anoxia (Figure 19).

Vertical and longitudinal patterns observed in nitrate-nitrite nitrogen on 10 September 1984 strongly support the contention that Hartwell Lake discharges move through Richard B. Russell Lake at the depth of the seasonal thermocline (Figure 21). Nitrate-nitrite nitrogen concentrations increased within a narrow layer of water located between the 6- and 12-m depths in the forebay area, and values were uniform from the dam to the headwater region. Above and below this zone, concentrations were undetectable.

Beaverdam Creek and Rocky River stations displayed differing seasonal patterns in the distribution of nitrogen during the stratified period. At the bottom depths of both embayment stations, total nitrogen and dissolved nitrogen exhibited an early summer increase in concentration at the bottom depth shortly after the establishment of thermal stratification (Figure 22). This pattern indicates that concentrations in the hypolimnion increased more rapidly in the arms than in the main basin. Ammonia-nitrogen increased to a similar maximum peak at Station 130 during this period; however, Station 140 did not display this trend (Figure 23). Concentrations of ammonia-nitrogen declined at both stations in June and July, then values increased in September. Nitrate-nitrite nitrogen was detectable in the hypolimnion of both stations in February, but values declined to undetectable levels during thermal stratification.

Phosphorus is an important nutrient for algal growth, and its pattern of distribution was influenced by interactions between inundated soils and hypolimnetic waters. Total phosphorus and soluble reactive phosphorus displayed concentration increases at the bottom depths during the stratified period, particularly from the forebay area to Station 160 (Figure 24). In September, total phosphorus concentrations at the bottom ranged from 0.169 mg P/l at Station 60 to 0.102 mg P/l at Station 160. Soluble reactive phosphorus exhibited a similar longitudinal concentration gradient, ranging from 0.150 mg P/l at Station 60 to 0.081 mg P/l at Station 160. Soluble reactive phosphorus concentrations were high in the hypolimnion during the stratified period and comprised a large percent of the total phosphorus.

Hypolimnetic phosphorus dynamics are often associated with iron and dissolved oxygen conditions. Bound phosphates are released from sediments when

Figure 22.   Temporal changes in nitrogen at the bottom depth
of Stations 130 (solid line) and 140 (broken line)
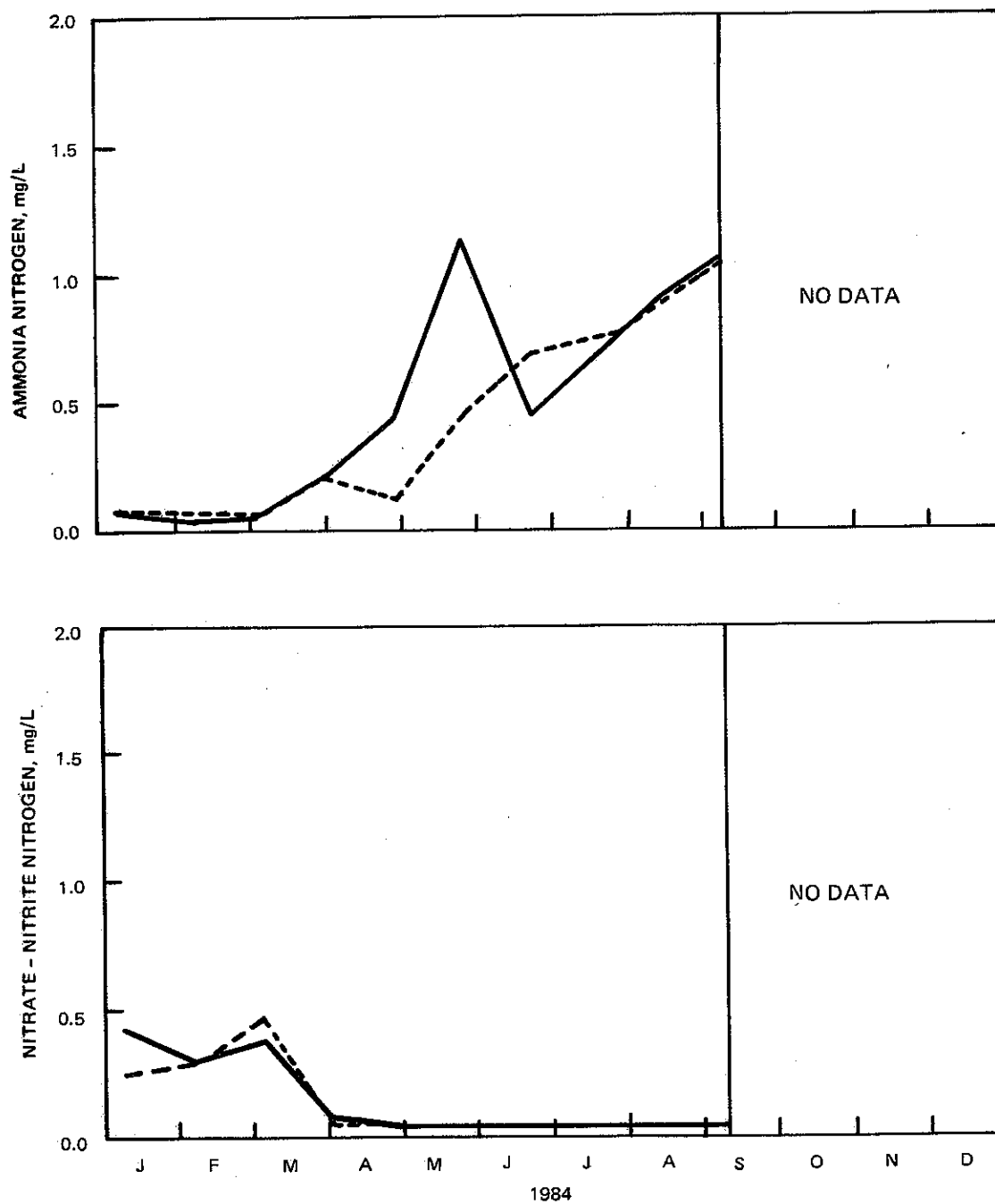
Figure 23. Temporal changes in ammonia-nitrogen and nitrate-nitrite nitrogen at the bottom depth of Stations 130 (solid line) and 140 (broken line)

Figure 24. Temporal and longitudinal patterns in total and soluble reactive phosphorus (mg P/l) at the 1-m depth (upper panels) and at the bottom depth (lower panels) for the main basin of Richard B. Russell Lake

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ferric iron is reduced to ferrous iron. Seasonal and spatial zones of soluble reactive phosphorus increase coincided with the occurrence of anoxia, suggesting that sediment phosphorus release was important in hypolimnetic phosphorus cycling.

Total phosphorus and soluble reactive phosphorus exhibited accumulation at hypolimnetic depths in the main basin on 10 September (Figure 25). Vertical total phosphorus concentration gradients ranged from 0.169 mg P/l at the bottom depth to 0.041 mg P/l near the 20-m depth in the forebay area. The

Figure 25. Vertical and longitudinal patterns in phosphorus for the main basin of Richard B. Russell Lake on 10 September 1984

extent to which accumulation occurred in the deep areas of the basin is indicated by the 0.050 mg P/l contour line in Figure 25. Above the depth of this contour, concentrations decreased and were longitudinally uniform. Both total phosphorus and soluble reactive phosphorus concentrations were low in the thermocline region and comparable to concentrations measured for Hartwell discharges. However, patterns suggestive of interflows could not be distinguished with these data.

In the surface waters, total phosphorus and soluble reactive phosphorus exhibited low concentrations throughout the stratified period (Figure 24). Total phosphorus concentrations were less than 0.005 mg P/l from May until September and uniform from the forebay area to Station 180. Above Station 180, total phosphorus concentrations appeared to be influenced by Hartwell Dam releases and exhibited a maximum of 0.050 mg P/l from May through August. Surface soluble reactive phosphorus concentrations were undetectable during a major portion of the stratified period due to assimilation by algae. Slightly elevated concentrations were detected in the lower area of the reservoir, and a peak was observed in the headwater region in June and July.

Phosphorus concentrations in the two embayment stations exhibited different patterns during the stratified period (Figure 26). As with other variables, total phosphorus and soluble reactive phosphorus increases were rapid in the bottom waters of these stations shortly after the onset of thermal stratification. This pattern coincided with the development of anoxia. Both forms of phosphorus exhibited a concentration peak in April and May, declines in June and July, and increases in September. Vertical trends in the water column were similar to patterns observed in the main basin. Total phosphorus concentrations were low in the epilimnion and metalimnion, and values rapidly increased in the hypolimnion. On 10 September, hypolimnetic concentrations of total phosphorus at Station 140 ranged from 0.196 mg P/l near the sediment surface (i.e., the 23-m depth) to 0.014 mg P/l at the 14-m depth. Soluble reactive phosphorus, which ranged from 0.126 mg P/l to 0.005 mg P/l at these respective depths, represented a substantial amount of the total phosphorus.

Concentration increases in the hypolimnion for iron and manganese appeared to be related to the occurrence of anoxia (Figures 27 and 28). Iron and manganese dynamics are closely associated with the occurrence of anoxia and a reducing environment. Both elements exhibit a change in valence state and become soluble as the redox potential falls. In addition, the inundated

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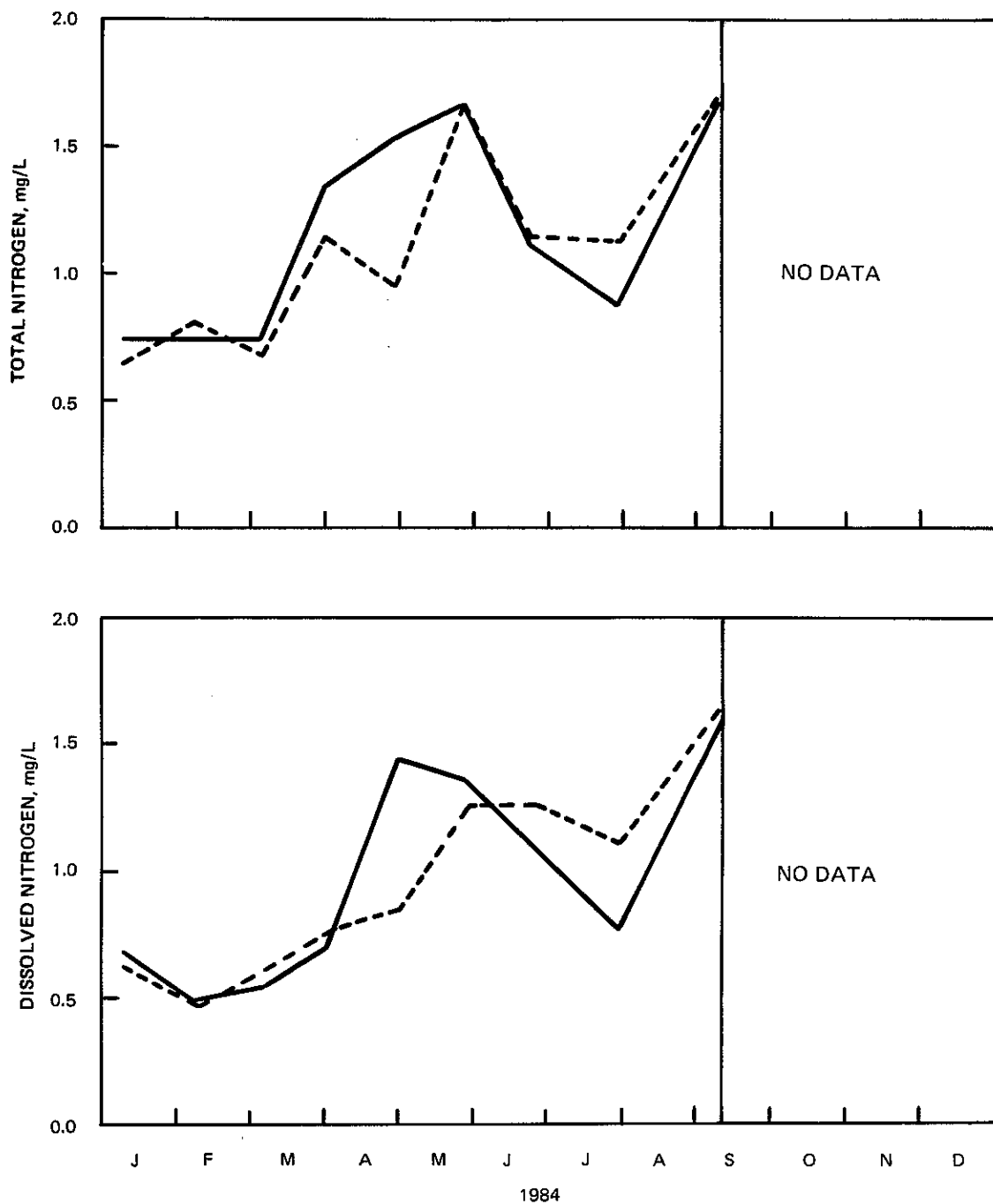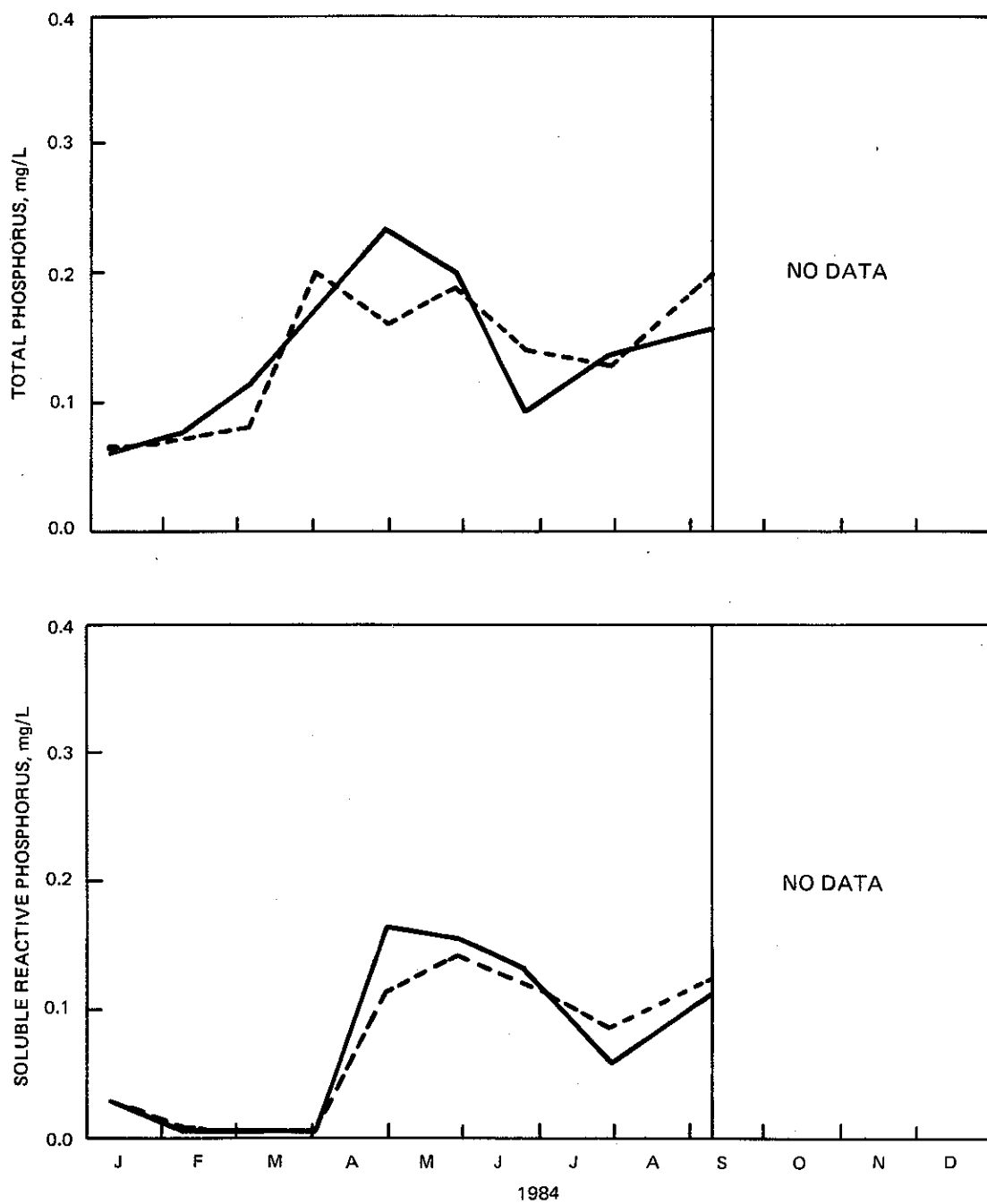
Figure 26. Temporal changes in phosphorus at the bottom
depth of Stations 130 (solid line) and 140 (broken line)
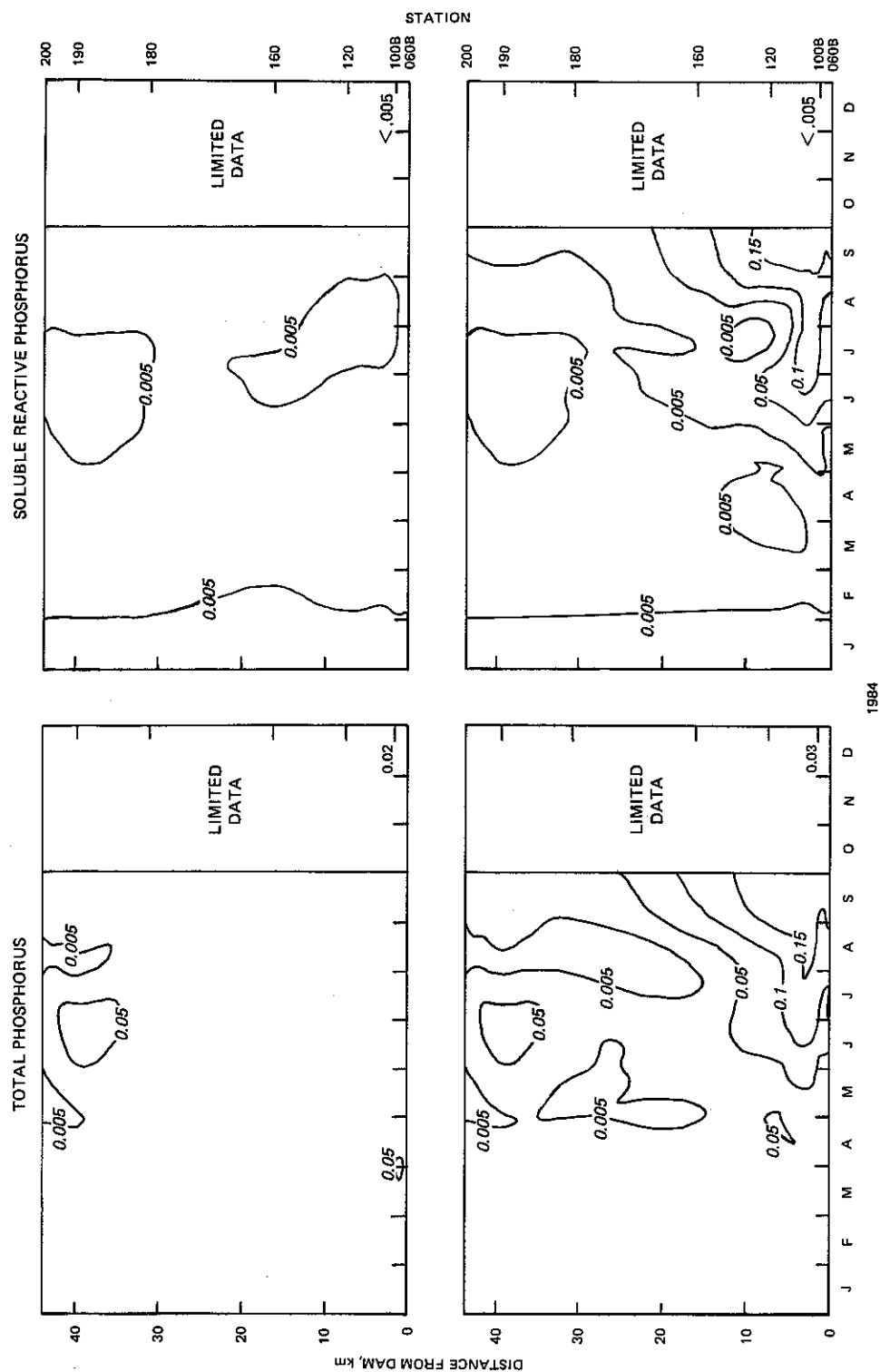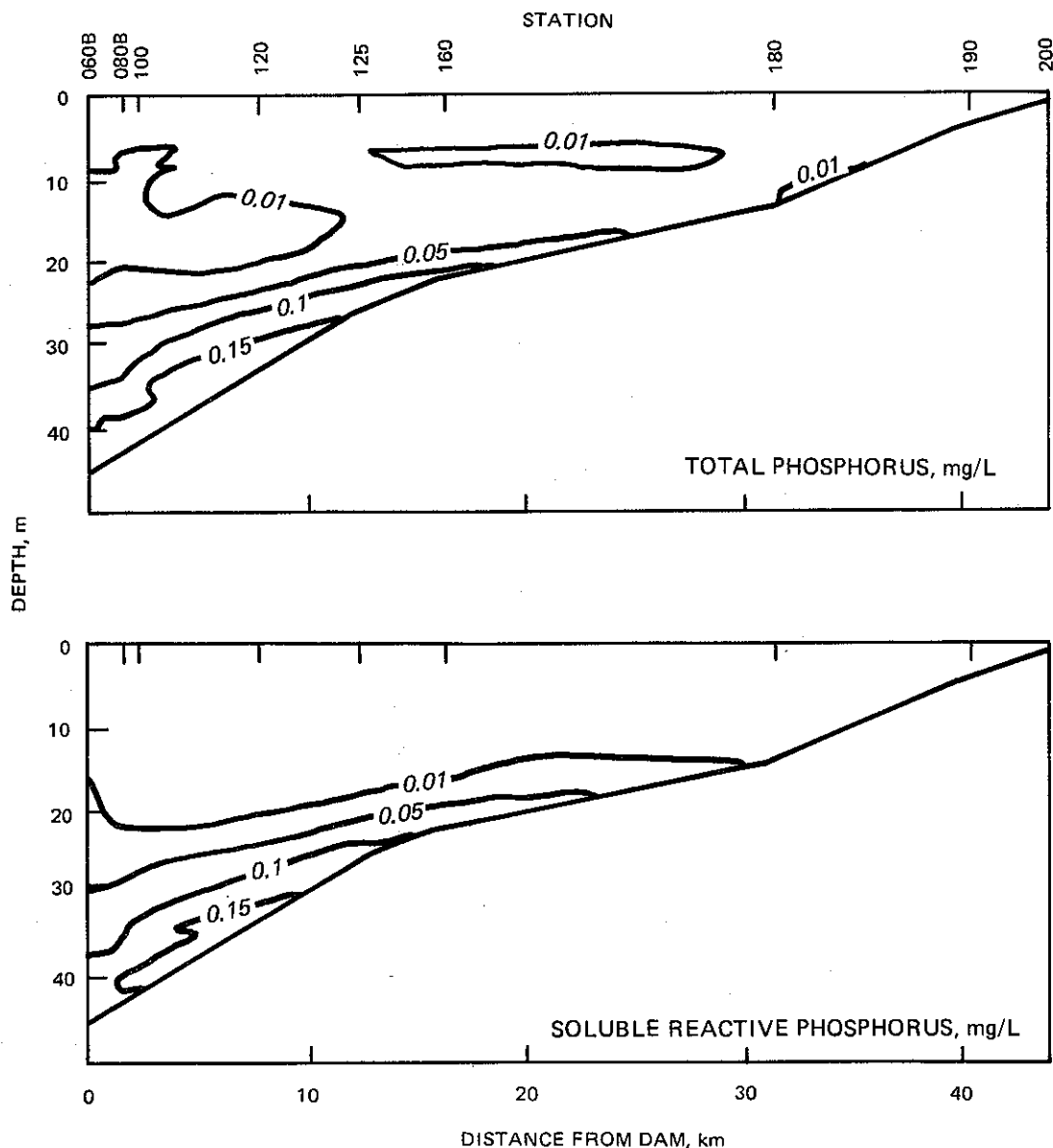
Figure 27. Temporal and longitudinal patterns in total and dissolved iron (mg Fe/l) at the 1-m depth (upper panels) and at the bottom depth (lower panels) for the main basin of Richard B. Russell Lake
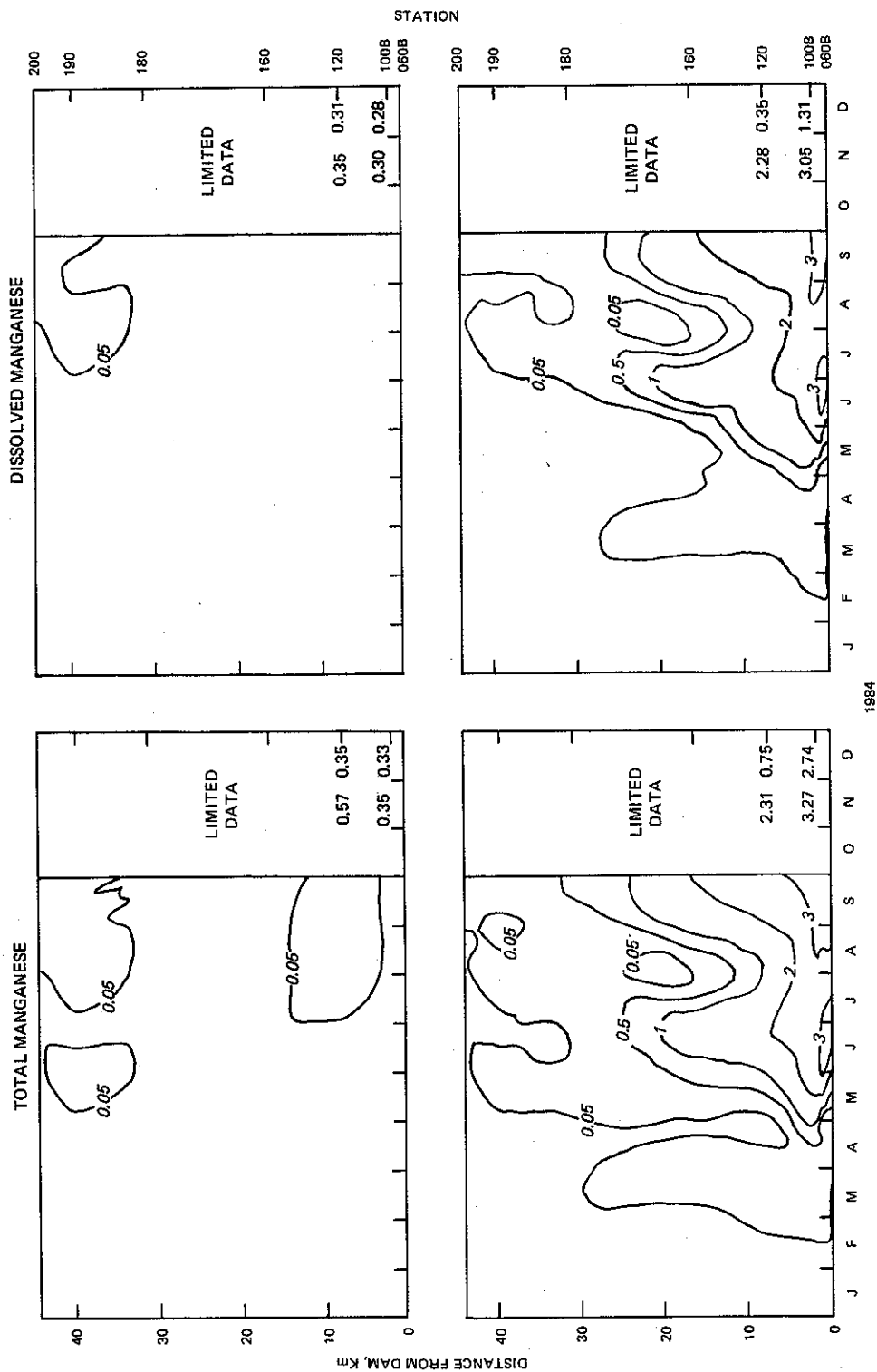
Figure 28. Temporal and longitudinal patterns in total and dissolved manganese (mg Mn/l) at the 1-m depth (upper panels) and at the bottom depth (lower panels) for the main basin of Richard B. Russell Lake

bottomlands appeared to be an important source, as both forms displayed pronounced concentration increases at bottom depths in the main basin during the stratified period. Elevated concentrations of both elements were observed along much of the length of the main basin. A large proportion of the total iron and total manganese was in the soluble form, as indicated by similar trends observed for total and dissolved iron and total and dissolved manganese.

At the surface depths, total and dissolved iron and manganese concentrations were low and did not exhibit spatial differences during the stratified period. This was expected since both elements form precipitates in the presence of dissolved oxygen. In the lower two-thirds of the main basin, total iron concentration (Figure 27) was less than 0.5 mg Fe/1, and total manganese (Figure 28) was undetectable from May until September. Values were slightly elevated in the headwater region, suggestive of Hartwell Dam influences. However, dissolved iron and manganese concentrations were generally undetectable during much of the stratified period.

The importance of inundated bottomlands as a source of iron and manganese was suggested by longitudinal and vertical patterns observed on 10 September (Figures 29 and 30). Total iron and manganese exhibited pronounced concentration increases in the hypolimnion on this date in areas exhibiting anoxia. Total iron values ranged from over 10.4 mg Fe/1 at Station 60 to 7.0 mg Fe/1 at Station 160 in the bottom waters. Total manganese exhibited a similar longitudinal concentration gradient with concentrations ranging from 3.8 mg Mn/1 at Station 60 to 1.7 mg Mn/1 at Station 160. Vertical concentration gradients were observed from Station 60 to Station 160. The extent to which concentrations increased in the water column is indicated by the 2- and 1-mg/1 contour lines for total iron and total manganese, respectively, in Figures 29 and 30. Above the 20-m depth, concentrations were lower and longitudinally uniform. Dissolved iron and dissolved manganese displayed similar concentration patterns in the hypolimnion and comprised a large percentage of the total concentration.

Total iron and manganese concentrations began increasing in the hypolimnion of the arms more rapidly than in the main basin. By April and May, total iron concentrations at the bottom (Figure 31) were 11.2 mg Fe/1 and 9.0 mg Fe/1 at Stations 130 and 140, respectively, while Station 60 exhibited a concentration of only 1.1 mg Mn/1. A similar pattern was observed for total

Figure 29. Vertical and longitudinal patterns in iron for the main basin of Richard B. Russell Lake on 10 September 1984
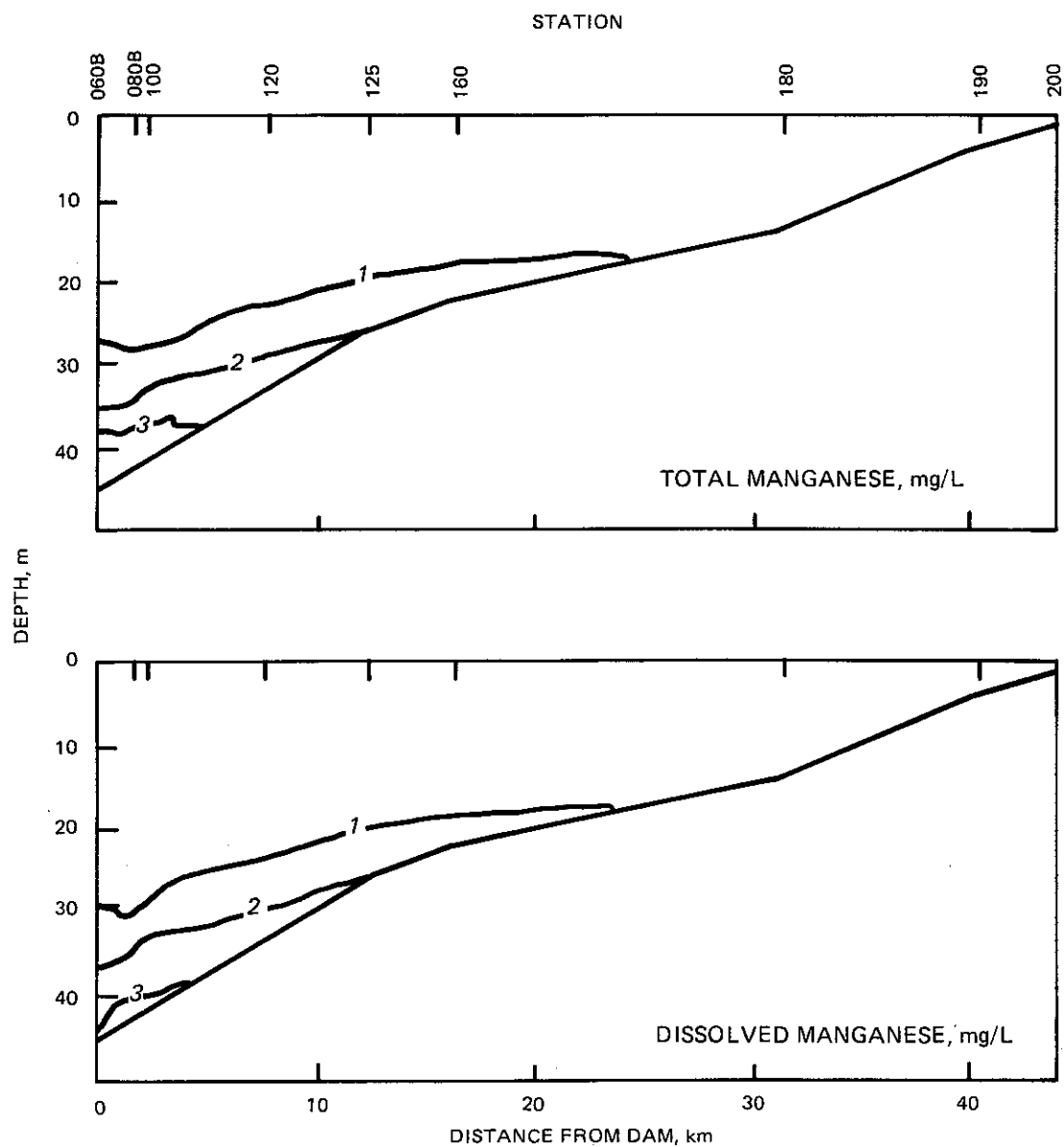
Figure 30. Vertical and longitudinal patterns in manganese for the main basin of Richard B. Russell Lake on 10 September 1984
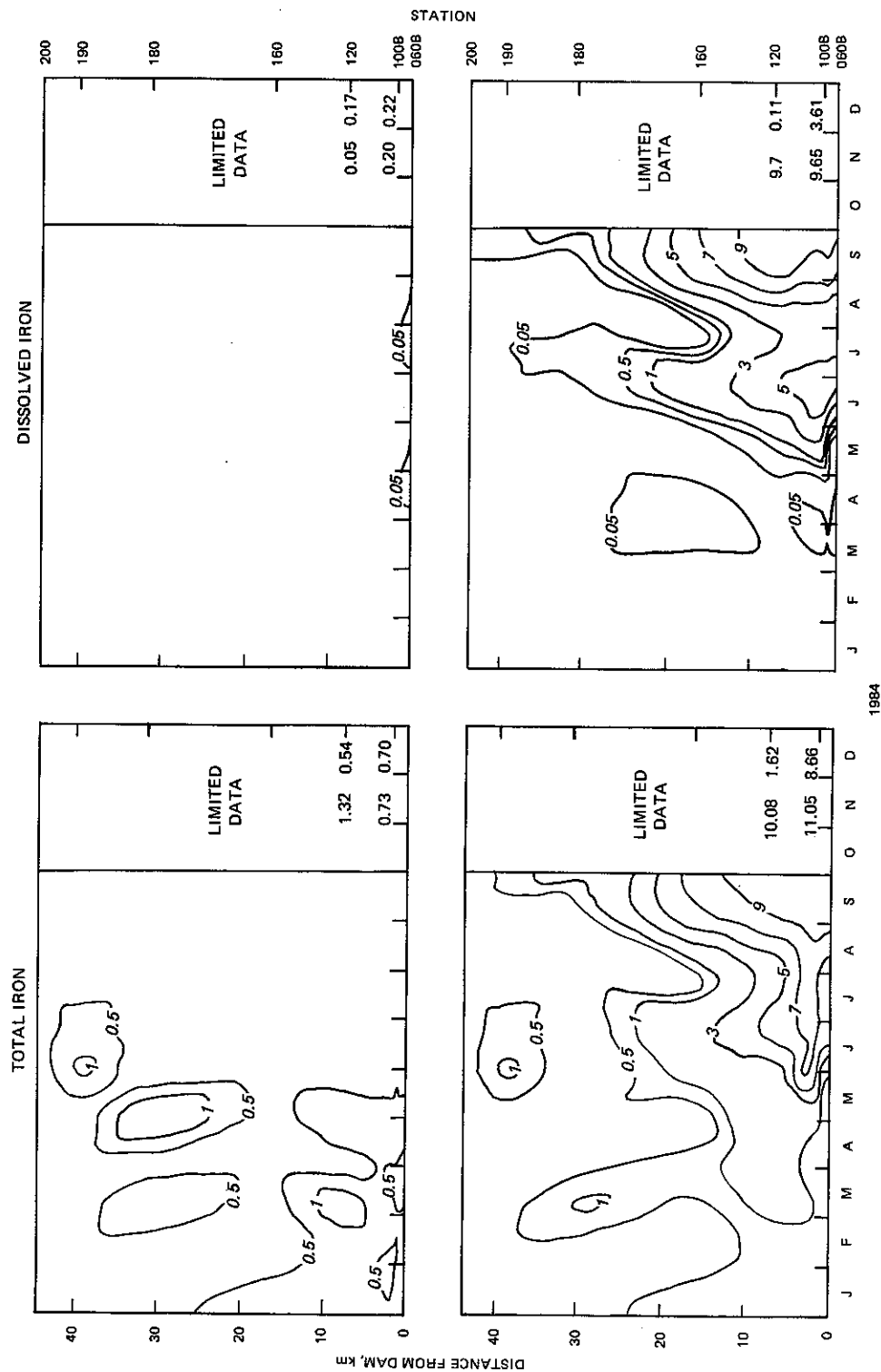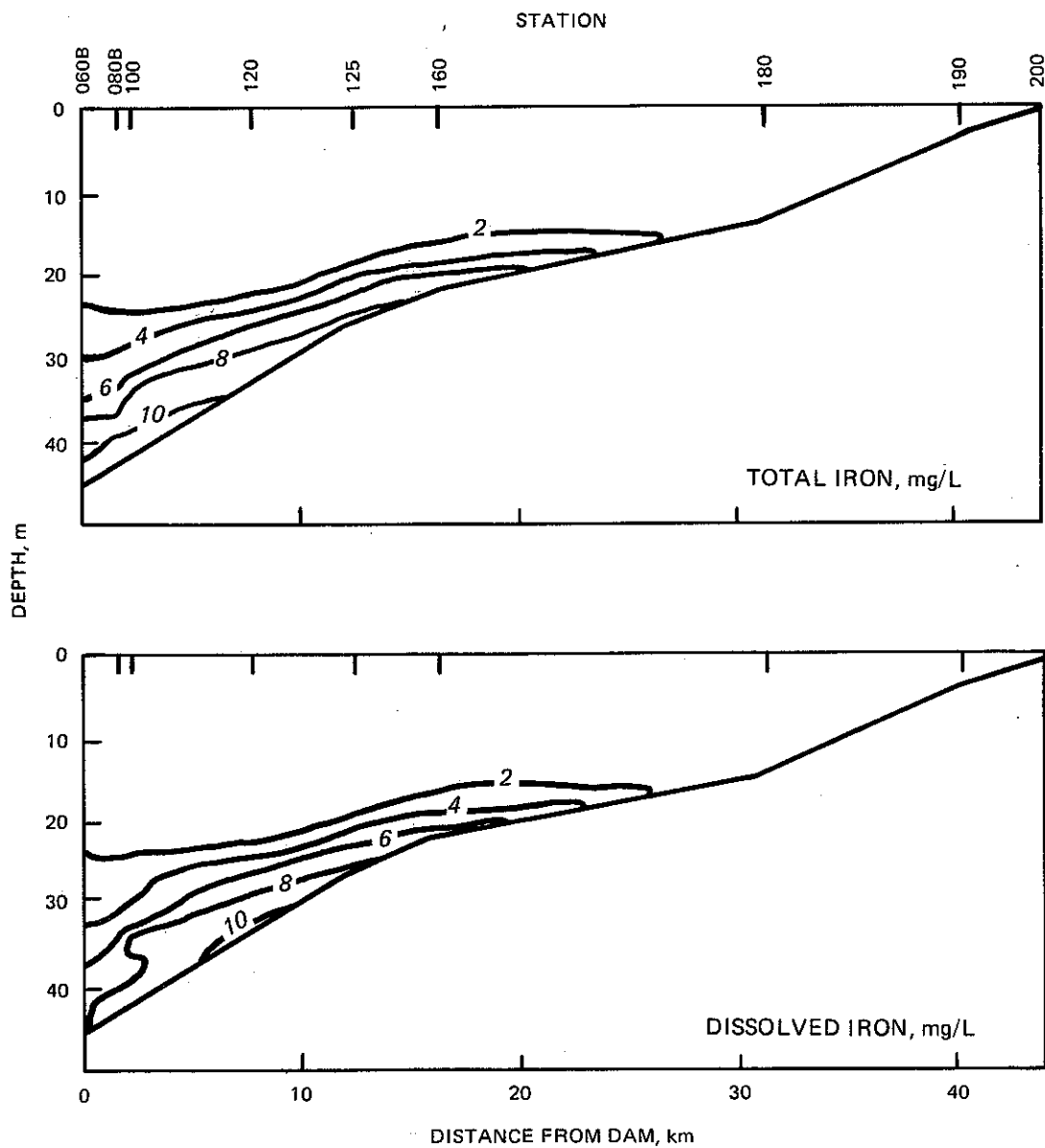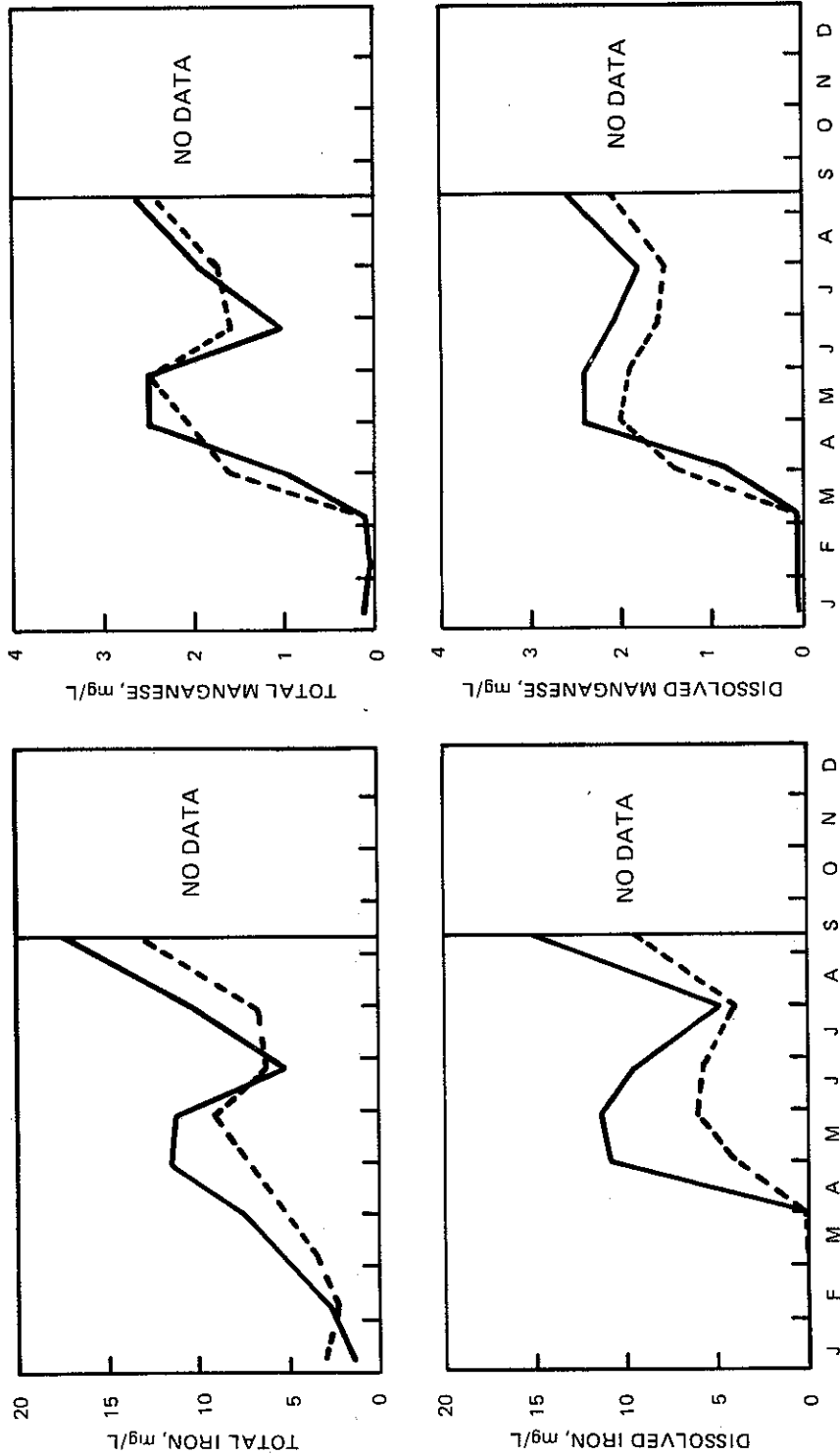
Figure 31. Temporal changes in total and dissolved iron and manganese at
the bottom depth of Stations 130 (solid line) and 140 (broken line)

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manganese (Figure 31). A seasonal trend of bimodal concentration maxima was observed for total manganese and total iron at both stations. After a concentration decline in June and July, values for both forms again increased to maxima in September. Most of the iron and manganese was in the soluble form in the hypolimnion during the stratified period.

Total calcium, sodium, magnesium, and potassium generally comprise a substantial proportion of the dissolved solids in fresh waters. These elements are important for algal growth and determine the degree of water hardness. Concentrations of these constituents were elevated in both the epilimnion and hypolimnion on 10 September (Figure 32). In the epilimnion, mean concentrations in the forebay area were 2.5 mg Na/l for total sodium, 2.1 mg Ca/l for total calcium, 1.5 mg K/l for total potassium, and 1.1 mg Mg/l for total magnesium. In the hypolimnion, vertical concentration gradients were evident from the bottom depth to near the 20-m depth. Total calcium exhibited the most pronounced concentration gradients, followed by total magnesium and total potassium. For example, in the forebay area total calcium exhibited a vertical concentration range of 4.5 mg Ca/l at the bottom to 1.5 mg Ca/l at the 20-m depth, while total sodium ranged from 2.4 to 2.9 mg Na/l at these respective depths. Total sodium displayed minimal vertical fluctuations in concentration during this period.

Uniform concentrations were observed for each of the above elements in the thermocline region along the length of the main basin. Concentrations were lower in this stratum than values observed in the epilimnion and hypolimnion, and comparable to values observed at Station 200 (Figure 32). For example, concentrations of total sodium at this depth interval ranged from 1.9 to 2.2 mg Na/l in the main basin, while Station 200 exhibited a concentration of 2.1 mg Na/l. These trends suggest the influence of Hartwell Dam discharges on chemical conditions in Richard B. Russell Lake.

Alkalinity and hardness are a measure of the buffering capacity and the polyvalent metallic ionic strength of water, respectively. Alkalinity is an indirect measure of carbonates and bicarbonates which increase the pH of water above 4.5 units, while hardness is based on the concentrations of calcium and magnesium. General patterns indicated a low buffering capacity and soft water for Richard B. Russell Lake during the stratified period. During stratification and anoxia, alkalinity and hardness exhibited modest increases in the hypolimnion. These increases were first detected in the two arms and later
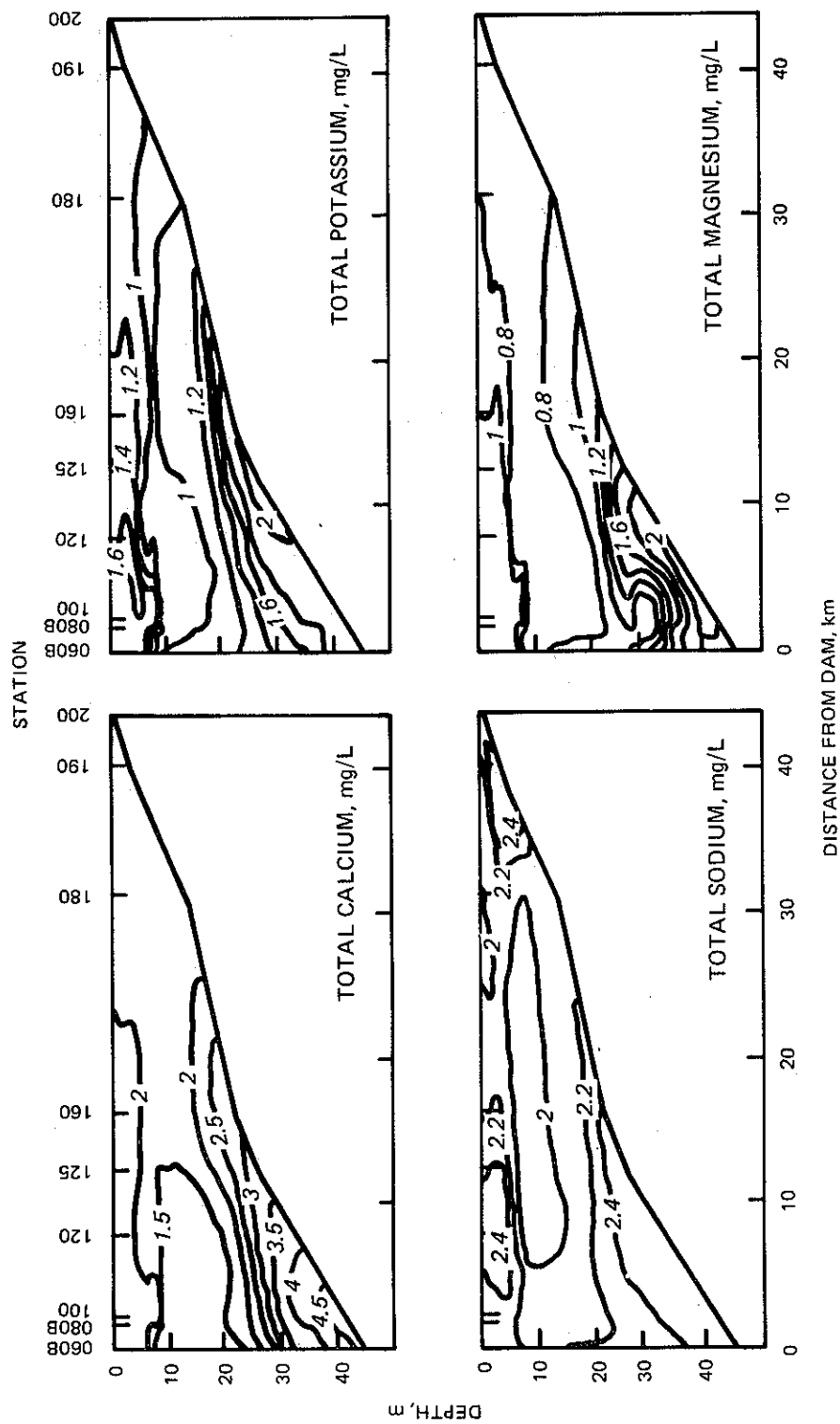
Figure 32. Vertical and longitudinal patterns in total calcium, total potassium, total sodium, and total magnesium for the main basin of Richard B. Russell Lake on 10 September 1984

in the main basin in June. On 25 June, alkalinity at the bottom depths ranged from 34 to 18 mg $CaCO_3$/1 for Stations 60 and 160, respectively. Stations 130 and 140 exhibited concentrations of 49 and 35 mg $CaCO_3$/1, respectively. Hardness exhibited similar spatial patterns, ranging from 13.3 in the forebay region to 15.5 at Station 130. Seasonal hypolimnetic increases in hardness reflected increases observed for magnesium and calcium.

Distinct vertical and spatial trends were observed on 10 September (Figure 33). On this date, concentration increases in alkalinity and hardness
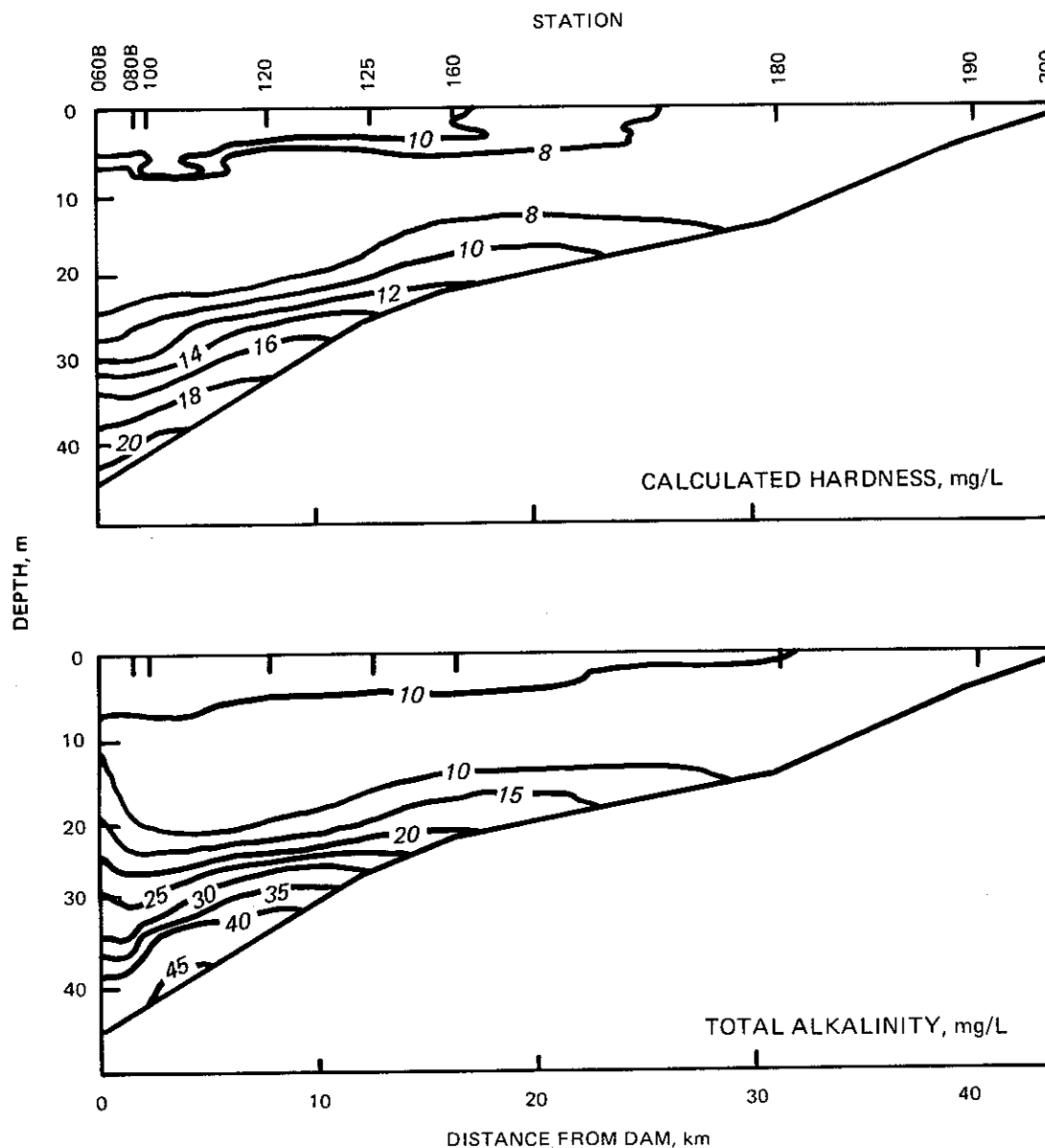
Figure 33. Vertical and longitudinal patterns in calculated hardness and total alkalinity for the main basin of Richard B. Russell Lake on 10 September 1984

were detected in the hypolimnion. In the metalimnetic region, and in the headwater region, concentrations were lower and uniform.

Sulfate concentrations were seasonally low and exhibited moderate spatial patterns in Richard B. Russell Lake. Concentration ranged from only 1.0 to 2.2 mg SO_4/l during the early spring months, then increased slightly in the epilimnion during the stratified period. Because bacterial reduction of sulfate occurs in an anaerobic environment, concentrations declined to undetectable levels in hypolimnetic areas experiencing anoxia during summer stratification. Associated with this decrease was an increase in sulfide in the anoxic hypolimnion. Although concentrations could not be determined with current analytical techniques, a strong, characteristic odor was detected when hypolimnetic water samples were collected. This odor was also detected in the Russell Dam releases late in the stratified period.

Secchi disc depth, an indirect measure of light transparency, exhibited small fluctuations during the study period. Secchi disc depth ranged from 0.5 to 2.5 m (Figure 34). Lowest Secchi disc depths occurred in the upstream portion of Russell Lake during the winter period. Increased turbidity associated with storm inflows, Hartwell Dam releases, and inflows from small tributaries may have reduced light transparency. During the stratified months, Secchi disc depth was slightly higher and longitudinally uniform.

Chlorophyll a, an indicator of algal biomass, was measured from integrated samples taken from the euphotic zone (i.e., two times Secchi disc). Chlorophyll a concentrations exhibited lowest values during the filling period and higher, more pronounced, concentration patterns during the summer stratified period (Figure 34). From January until June, values were low and ranged from less than 2 to 5 mg/cu m. These concentrations were longitudinally uniform from Station 60 to Station 160.

A lakewide concentration increase occurred as surface waters warmed from July to August. During this period, inflows from Hartwell Dam appeared to have an influence on the distribution of chlorophyll a. Highest concentration was observed near Station 180 (i.e., <14 mg/cu m) with a trend of decreasing values toward Russell Dam. In the vicinity of Station 180, inflows from Hartwell Dam plunge and continue as a density current. Therefore, the continual input of nutrients to this area may have stimulated algal productivity. Concentrations upstream of Station 180 (i.e., upstream from the plunge point) were seasonally lower during the stratified period and may have been

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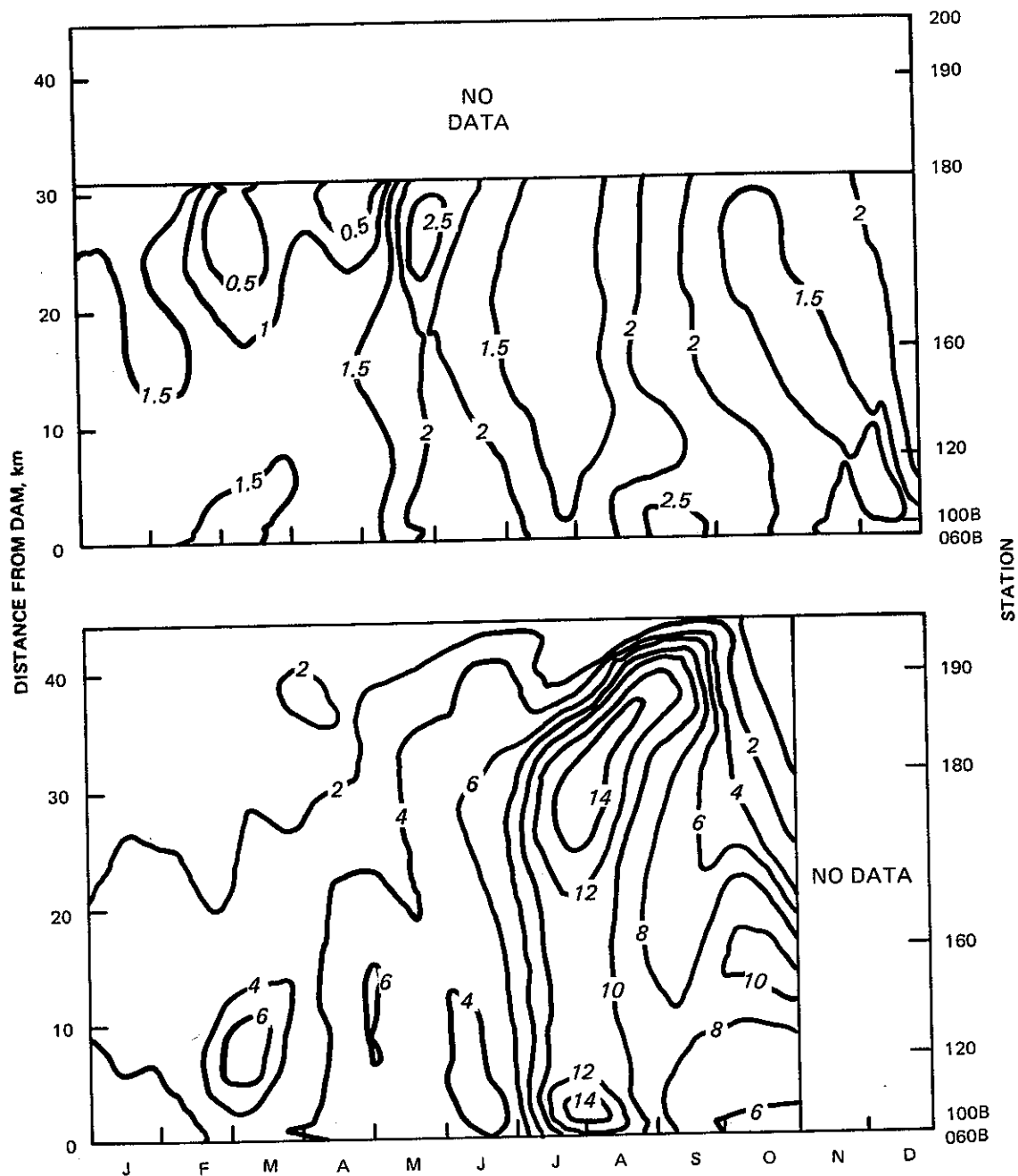
Figure 34. Temporal and longitudinal patterns in Secchi disc depth (meters; upper panel) and integrated chlorophyll a concentrations (mg/cu m; lower panel) for the main basin of Richard B. Russell Lake

influenced by the high inflow velocities in the Hartwell Dam tailwater area, which would tend to move algal populations down-reservoir. This pattern of lower concentrations was observed from Station 200 to Station 180 in October. In the lower two-thirds of the lake, chlorophyll a concentrations exhibited a peak in the forebay area in late July; however, concentrations were

generally uniform throughout the stratified period.

A more detailed discussion of phytoplankton distribution and productivity is presented in Kimmel et al. (1985) and Hains (1985). In general, Kimmel et al. reported that primary productivity and chlorophyll a concentrations were highest in July and decreased in September. Pronounced longitudinal trends of increasing levels toward Hartwell Dam were observed in July. Hains found that periphytic biomass, which occurs on submerged structures (i.e., inundated trees), reached a maximum peak in April, then declined to minimal levels throughout the remainder of the stratified period. This was contrary to the expectation that periphyton would be an important component in the total primary-producer biomass of Russell Lake. Hains reported that anoxic conditions at the depth of the submerged treeline may have inhibited periphytic growth during summer stratification.

Limnological conditions during mixing. Autumnal mixing was a critical event with respect to water quality conditions in Richard B. Russell Lake. Throughout the summer (stratified) months, anoxic conditions occurred in much of the water column in the main basin and in the two major tributary embayments, and exchanges between inundated soils and bottom water led to the accumulation of nutrients and metals in the hypolimnion. Therefore, water quality problems existed when exchanges between the epilimnion and hypolimnion occurred during mixing. Since mixing resulted in a redistribution of anoxic water high in concentrations of readily oxidized materials (i.e., organic carbon and reduced metals), the depression of dissolved oxygen concentrations was a concern.

Mixing was strongly influenced by weather conditions and basin morphometry. Seasonally cooler air temperatures led to heat loss from the surface waters and the initiation of mixing by late August. Establishment of isothermal conditions occurred in late December. Since mixing resulted in a progressive deepening of the thermocline, near-isothermal conditions were first observed in the upstream portion of the main basin and in embayments where basin morphometry was shallow.

By 2 October, surface temperatures had decreased from 28 deg. C (in June) to near 20 deg. C in much of the main basin (Figure 35). Although a thermocline was observed in the forebay area, the metalimnetic region did not exhibit pronounced temperature gradients with increasing depth. Station 160, for example, displayed a temperature range of 21.1 deg. C to 15.6 deg. C from the surface to the bottom.
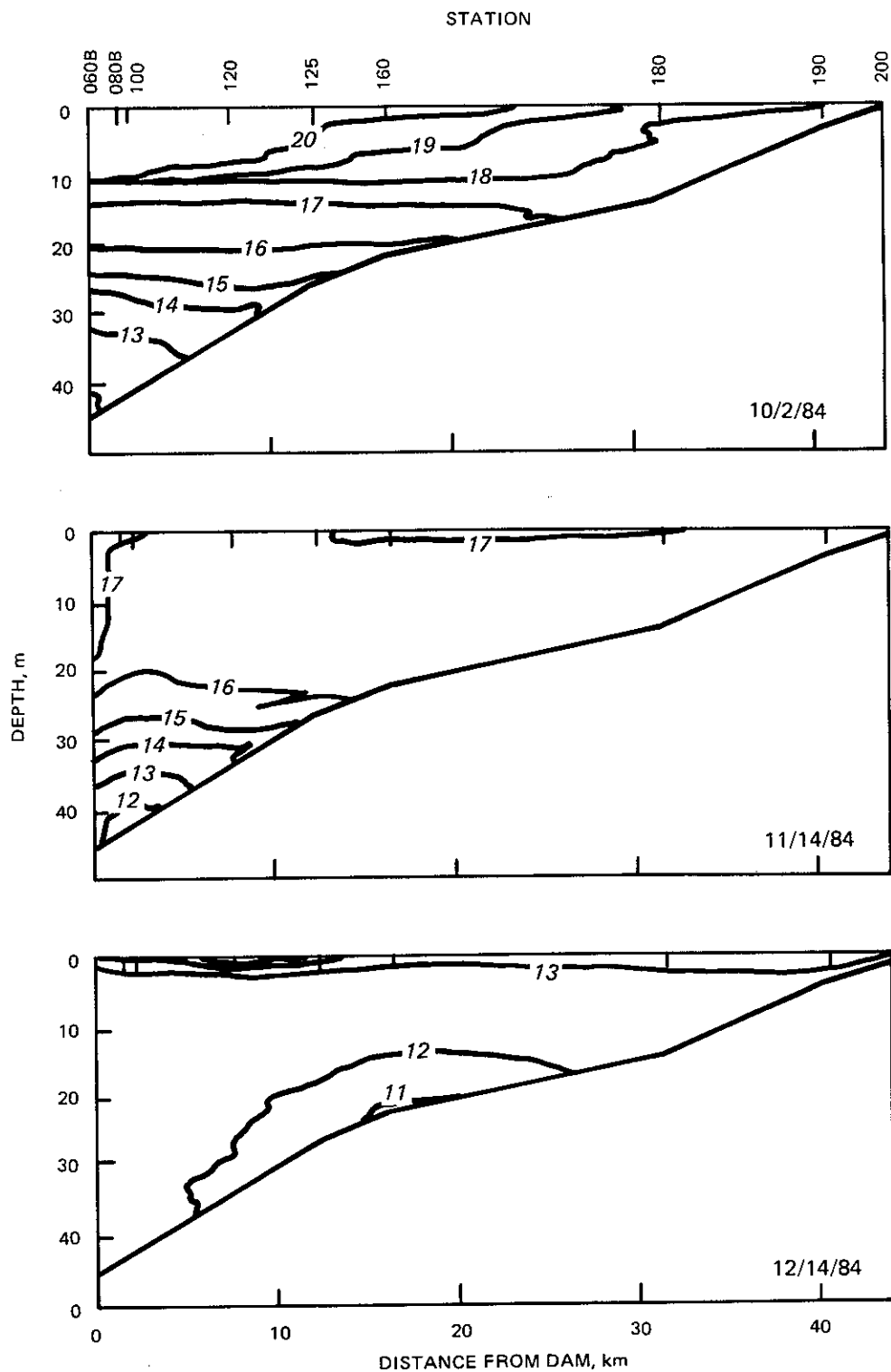
Figure 35. Vertical and longitudinal patterns
in temperature (deg. C) for the main basin of
Richard B. Russell Lake

Weak thermal stratification and mixing were evident on 14 November. Temperature on this date declined to 17 deg. C in the epilimnion and was nearly uniform to a depth of 20 m (Figure 35). Station 160 and the embayment stations (i.e., Stations 130 and 140) less than 20 m in depth were, therefore, isothermal by November. In comparison, Station 60 remained weakly stratified and had a temperature range of 17.6 to 12.1 deg. C. Further water column cooling and mixing resulted in near-isothermal conditions throughout the reservoir on 14 December (Figure 35).

Severe dissolved oxygen conditions were experienced throughout the lake during the mixing period. These conditions, which potentially threatened the Russell Lake fishery and downstream areas, were the direct result of the redistribution of reduced materials previously stored in deep, anoxic strata during the summer stratified period. These materials, when mixed to shallower strata, exerted an immediate and significant demand on dissolved oxygen stores. Throughout much of the lake, these demands were such that reaeration of surface waters was insufficient to prevent concentrations from declining to dangerously low levels.

Significant spatial and vertical patterns were observed in dissolved oxygen concentrations from October until December. In October, Richard B. Russell Lake exhibited a large zone of hypolimnetic anoxia which extended from Station 60 to Station 160 (Figure 36). Upstream of Station 160, dissolved oxygen concentration was depressed (i.e., <5.0 mg/l) as a result of releases from Hartwell Dam containing low levels of dissolved oxygen. Expansion of the epilimnion by 14 November led to erosion of the anoxic zone to the 20-m depth and severely depressed dissolved oxygen levels in the upper water column (Figure 36). During this period, shallow areas of the reservoir experiencing complete water column mixing exhibited the lowest dissolved oxygen concentrations. For example, levels at Station 160 were less than 2.0 mg/l in over 90 percent of the water column, while levels at Station 60 were still greater than 5.0 mg/l in the upper water column. These differences are related to the fact that oxygen-demanding materials stored in the hypolimnion were completely redistributed in the water column of the shallower stations. On 14 November, Hartwell releases appeared to be entering Russell Lake as a near-bottom density current in the vicinity of Station 180. Within these strata, dissolved oxygen concentrations were higher.

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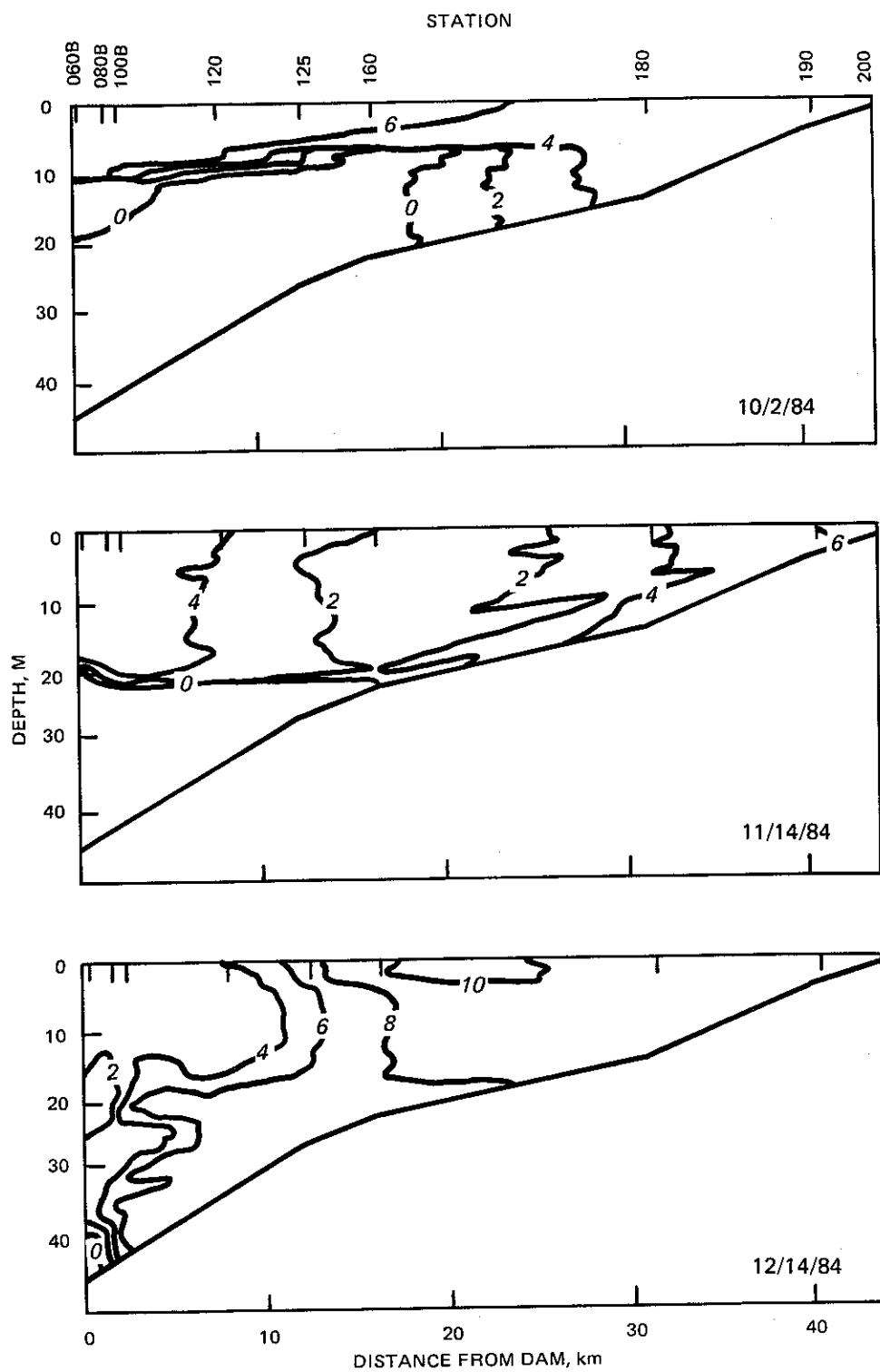
Figure 36. Vertical and longitudinal patterns in
dissolved oxygen (mg O_2/l) for the main basin of
Richard B. Russell Lake

During the final stages of mixing in the forebay area, the anoxic zone was significantly eroded, and dissolved oxygen concentrations were depressed in the water column (Figure 36). Upstream portions of the reservoir, which had experienced complete turnover 1 month earlier, exhibited a recovery of dissolved oxygen concentrations to near saturated levels. This pattern would be expected as reduced materials become depleted in the water column via oxidation and precipitation. Complete recovery of dissolved oxygen concentrations to saturated levels (i.e., 10 to 11 mg/l) was observed in the forebay area on 14 January 1985.

Also apparent from mid-November until the latter part of December was the progressive intrusion of water moderately high in dissolved oxygen (4 to 6 mg/l) between the 20- and 40-m depth in the forebay area. On 14 December, for instance, maxima in hypolimnetic dissolved oxygen were observed from Station 120 to Station 100 (Figure 36). Vertical profiles taken on 18 December more clearly displayed this trend (Figure 37). These patterns suggest the occurrence of near-bottom density currents, originating from Hartwell Dam discharges.

Associated with changes in temperature and dissolved oxygen at Station 60 was a redistribution in iron and manganese concentrations during the turnover period. Mixing and progressive erosion of the anoxic zone resulted in elevated concentrations of these forms in the upper water column. These increases were of immediate concern from an operational standpoint since releases to Clarks Hill Lake were from both tainter gates and penstocks. Introduction of dissolved oxygen to deep strata during the final stages of turnover led to marked decreases in metal concentrations throughout the water column via oxidation and precipitation.

Mixing events in November resulted in increased metal concentrations in the upper water column, which coincided with a concomitant depression in dissolved oxygen concentration. For example, mixing was apparent down to a depth of 30 m on 23 November as suggested by temperature and dissolved oxygen profiles taken at Station 60 (Figure 37). In the mixed layer, total iron and manganese increased from concentrations of 0.1 mg/l for both elements to approximately 0.7 mg Fe/l for total iron and 0.4 mg Mn/l for total manganese, suggesting an upward redistribution of these elements as mixing progressively eroded the anoxic zone. Below the zone of mixing, concentration gradients were still evident where anoxia was present. Levels at the bottom depth of
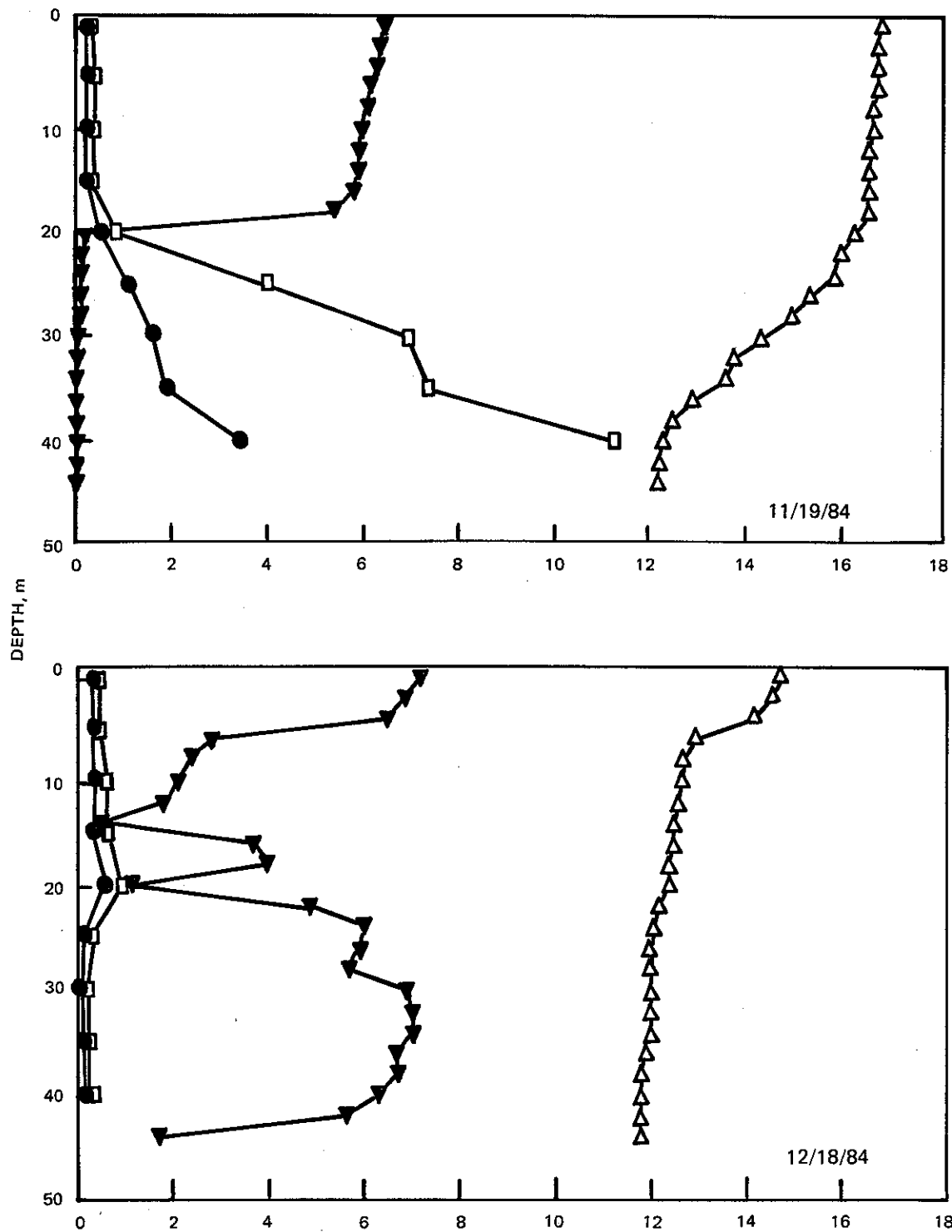
Figure 37. Vertical patterns in temperature (closed circle; deg. C), dissolved oxygen (closed triangle; mg O_2/l), total iron (square; mg Fe/l), and total manganese (open circle; mg Mn/l) at Station 60

Station 60 were over 10 mg Fe/1 for total iron and similar to values observed in September.

Coincident with the establishment of oxygenated conditions in early December was a marked reduction in total and dissolved forms of iron and manganese at Station 60. Dissolved concentrations at bottom depths decreased from over 10 mg Fe/1 to less than 0.5 mg Fe/1 for dissolved iron and from 2.0 mg Mn/1 to less than 0.5 mg Mn/1 for dissolved manganese (Figure 38).
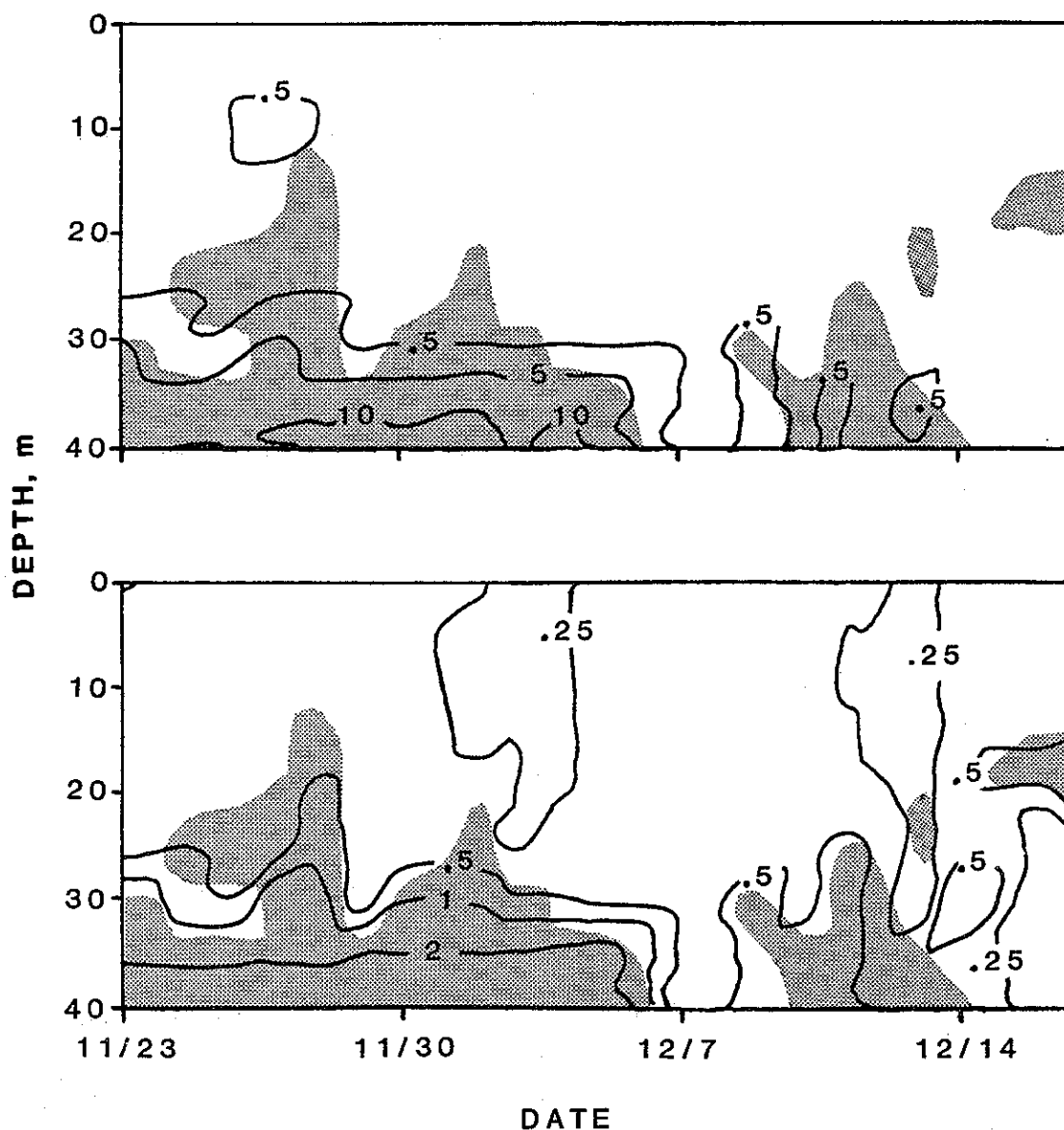
Figure 38. Temporal and vertical changes in dissolved iron (mg Fe/1; upper panel) and manganese (mg Mn/1; lower panel) at Station 60 in Richard B. Russell Lake from 23 November to 17 December 1984. Shaded areas indicate dissolved oxygen concentrations less than 1.0 mg O_2/1

Observations on 18 December indicated similar decreases in total iron and manganese (Figure 37).

During the turnover period, there were also significant hydrological changes. Periodic testing of the first generator unit was conducted in December, resulting in the release of water via penstocks from the mid-hypolimnion region. However, due to the poor water quality conditions of these strata (i.e., low dissolved oxygen, high nutrient and metal concentrations), surface releases were also employed in an effort to improve the discharges. As conditions worsened in Richard B. Russell Lake, releases were stopped to alleviate potentially adverse impacts on fish habitats and water quality conditions in the headwater region of Clarks Hill Lake. When dissolved oxygen and chemical concentrations improved in the forebay area in late December, generator testing was resumed.

Overview of seasonal patterns in in-situ variables at Station 60. Seasonal trends in thermal structure at Station 60 are depicted in Figure 39. In general, Station 60 exhibited near-isothermal conditions during January through mid-March, stratification from April to October, and turnover in December. During the stratified period, epilimnetic thickness was shallow, ranging from 3 m in June to 6 m in September, and marked temperature gradients were observed within a small vertical region in the metalimnion. As
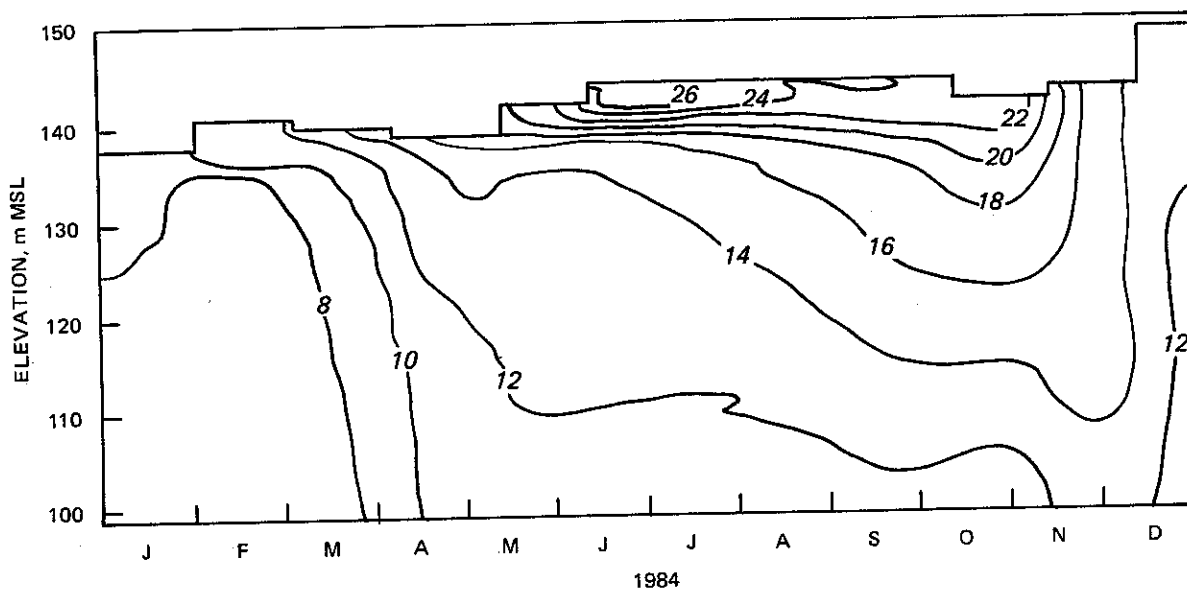
Figure 39. Temporal and vertical patterns in temperature (deg. C) at Station 60 during 1984

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discussed, this pattern may have been related to the discharge of surface water from tainter gates, which restricted epilimnetic expansion. The hypolimnion was expansive and exhibited minimal temperature increase throughout the stratified period. With no hypolimnetic withdrawal during the stratified period, the hypolimnetic residence time was high.

Dissolved oxygen was rapidly depleted in late spring, and a large zone of anoxia existed in the hypolimnion during the stratified period (Figure 40).
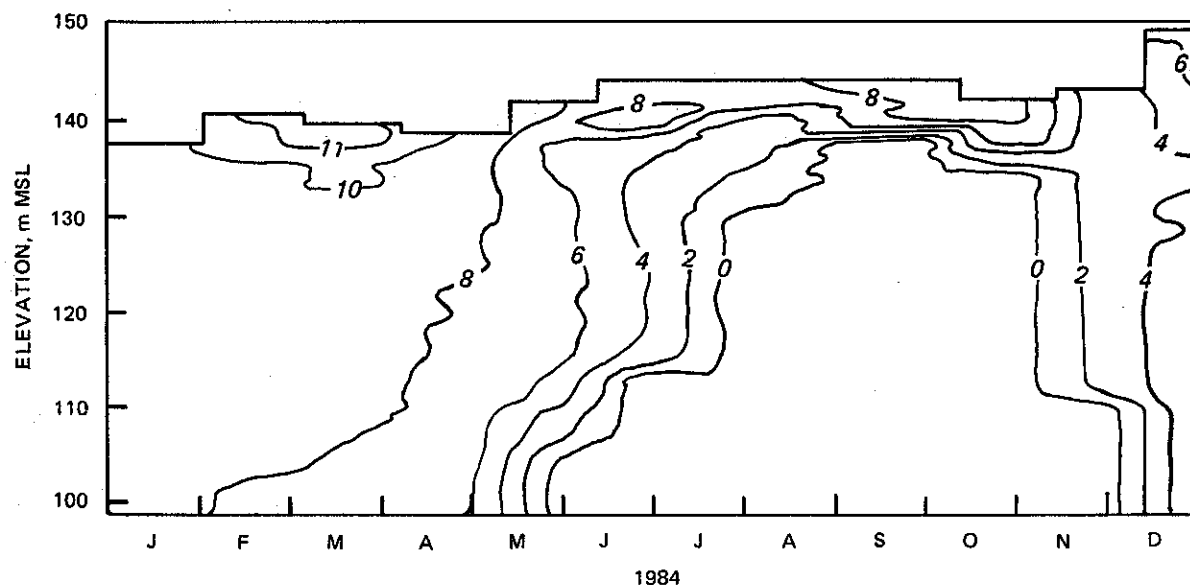
Figure 40. Temporal and vertical patterns in dissolved oxygen (mg O_2/l) at Station 60 during 1984

Hypolimnetic concentrations began decreasing in late-April, and anoxia was first detected in the bottom waters in early June. By September, anoxic conditions had developed from the bottom depth to the 10-m depth. Mixing and epilimnetic expansion progressively eroded the anoxic zone during November, and concentrations declined to less than 4.0 mg/l throughout much of the water column. Recovery of concentrations to greater than 6.0 mg/l occurred in January 1985.

Associated with anoxia was a shift in the oxidation-reduction potential to negative values and increased specific conductance values in the bottom waters (Figure 41). Negative redox potentials, which were first observed in bottom strata in early June during the development of anoxia, where observed high in the water column by September. Specific conductance values began increasing near the sediment surface shortly after the occurrence of anoxia and
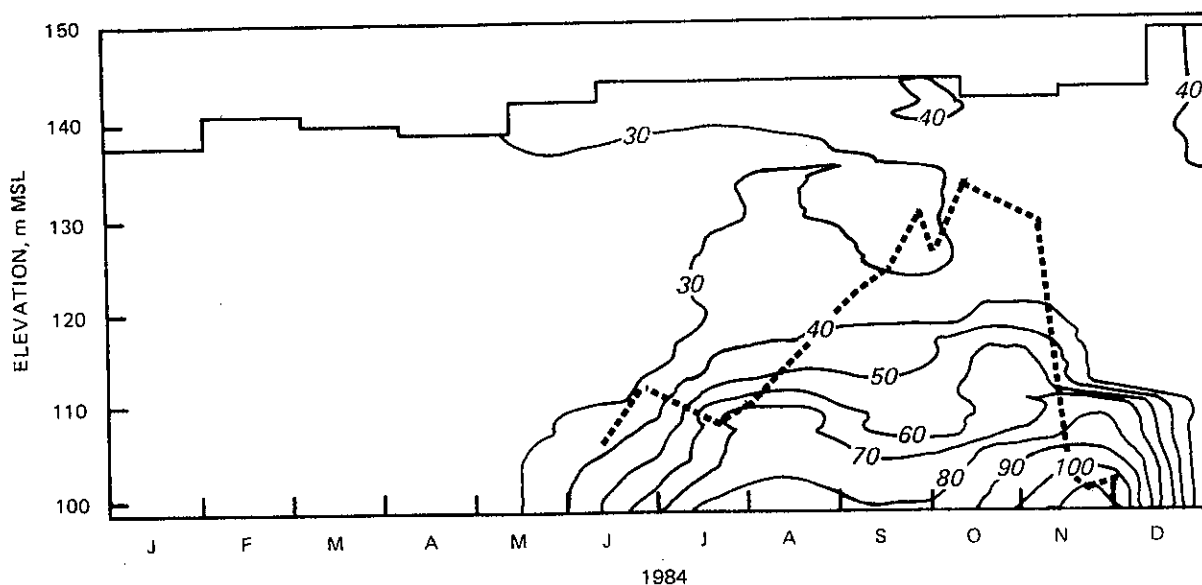
Figure 41. Temporal and vertical patterns in specific conductance
(umhos/cm) at Station 60 during 1984. Dashed line delineates area
where oxidation-reduction potential changes to a negative value

the shift to negative redox potentials. During the stratified period there
was a progressive increase in specific conductance values at the bottom depths
from June until November. Vertical gradients of decreasing values were evi-
dent toward the lake surface. These trends suggested the accumulation of dis-
sociated ions in the hypolimnion. After turnover, redox potentials were posi-
tive and specific conductance values decreased.

Hypolimnetic Oxygen Demands in Richard B. Russell Lake and Prediction Methods for Oxygen Injection Requirements

The oxygenation system for Richard B. Russell Lake was designed based
on oxygen demands calculated for Clarks Hill Lake, assuming an average daily
design discharge of 12,500 cfs and an oxygen absorption efficiency of 75 per-
cent. Maximum oxygen deficit was assumed to be 3.8 mg/l for the period
1-15 October, and the depletion rate was assumed to be 0.1 mg/l per day.
Since oxygen demands in a newly filled reservoir will be substantially greater
than in a mature reservoir such as Clarks Hill Lake, this section will discuss
the magnitude, effects, and implications of this greater demand on operation
of the oxygenation system.

For the purpose of this discussion, the oxygen deficit is defined as the

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difference in oxygen concentration from the design discharge concentration of 6 mg O_2/l. During the stratified period, any water moving downstream to the area of the oxygenation system may have a deficit of from 0 to 6 mg O_2/l which will have to be satisfied in order to meet the discharge standard. The oxygen debt includes the oxygen requirements of anoxic water, which must be satisfied in addition to the deficit. An oxygen debt is established when reduced materials (e.g., iron, manganese, sulfide, and ammonia) accumulate in the anoxic bottom water. Oxidation of these materials therefore results in an additional oxygen demand. While the deficit can be measured directly, measurement of the debt must be made indirectly. After the deficit and debt in the water approaching the diffuser have been satisfied, oxygen will continue to be utilized by various chemical and biological processes in the hypolimnion; the rate of this oxygen utilization is defined as the depletion rate.

The deficit, debt, and depletion rate are volume-based terms which can be calculated as a water column average for various regions of the reservoir and for each of the layers or strata of the water column. The calculations can also be made on an areal basis to relate the oxygen demand to the relative area of the hypolimnion in various areas of the reservoir. The depletion rate can be calculated from changes in successive oxygen profiles after the onset of thermal stratification and by assuming that mixing does not occur between the successive measurements. The rate can only be calculated from oxygen profiles until anoxic conditions occur; after this point, other estimation methods must be employed (e.g., biochemical oxygen demand).

Maximum depletion rates for various stations in Richard B. Russell Lake for the 1984 spring period were calculated from changes in successive in-situ oxygen profiles using the program PROFILE (Walker, 1985). Maximum water column averages are listed in Table 5; these values were calculated assuming an upper hypolimnion elevation of 125 m MSL and an upper metalimnetic elevation of 135 m MSL for all stations. Maximum values for each 2-m layer are listed in Table 6; at each station the maximum rate for each layer is listed along with the oxygen concentration at the beginning and end of the period of maximum oxygen depletion for that layer.

Maximum rates occurred in the Rocky River and Beaverdam Creek embayments (i.e., Stations 130 and 140) of the reservoir early in the season. Hypolimnetic values were greater than 0.6 mg/l per day at these stations from early to mid-March. Maximum rates in the main stem occurred somewhat later in the

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season and appeared to occur earliest near Station 120 and progress down to Station 60 near the dam and up to Station 160. Maximum rates at Stations 60 and 120 were generally between 0.1 and 0.3 mg/l per day while the maximum rates at Station 160 approached 0.4 mg/l per day. Vertical trends at particular stations over time reflected progressive movement of anoxic conditions upward in the water column. Once a particular layer reached zero milligrams per liter dissolved oxygen, the depletion rate could no longer be calculated. Depletion rate calculations were also complicated by advective mixing events which occurred during the spring period and may have introduced oxygenated surface water to deeper depths. This may have resulted in an underestimate of rates for some layers at certain times. Rates at Station 60 were low partly as a result of operational events since use of sluice gates at the dam may have permitted mixing of surface water down into the hypolimnion until usage of these gates ceased. Rates at Stations 130 and 140 were high as a result of greater sediment surface area in contact with the water column and also as a result of high oxygen-demanding water entering these areas. This is particularly true for the Rocky River embayment (Station 140), which receives discharges from the eutrophic Lake Secession.

Maximum areal depletion rates are shown in Table 7. These values were also calculated using the program PROFILE and by assuming top elevations for the hypolimnion and metalimnion at 125 and 135 m MSL, respectively, for all stations. Longitudinal trends are similar to those discussed for volumetric rates, with maximum values occurring in mid-March at Stations 130 and 140, progressing toward midreservoir at Station 120 and then to Stations 60 and 160 by mid-May.

Potential sources of oxygen demand in Richard B. Russell Lake have been extensively discussed by Gunnison et al. (1983, 1984), and a summary and discussion are presented here with respect to observations of 1984 trends. Inundation of soils and vegetation during reservoir filling resulted in the release of substantial amounts of oxygen-consuming materials and plant nutrients. Soil samples from the Richard B. Russell Lake basin had oxygen demands high enough to result in depletion of oxygen from the overlying water column in laboratory studies; the subsequent development of anoxic conditions in the water column led to release of large amounts of dissolved organic matter, plant nutrients, iron, manganese, and hydrogen sulfide. In Richard B. Russell Lake, water containing these materials will flow downstream toward the oxygen

injection system, and the oxygen demand of this water will have to be satisfied in order to meet the downstream water quality objective.

Gunnison et al. (1984) reported a biochemical oxygen demand (BOD) for leaf litter that was initially 4 to 20 times greater than that of soil, in a laboratory study performed at 20 deg. C. In addition, chemical oxygen demand (COD) greatly exceeded BOD. Initially, the 5-day BOD from vegetation was about 8 mg/l and declined to 0.92 mg/l after 50 days; initial COD was nearly 50 mg/l and declined to about 20 mg/l after the same period. Much of the COD resulted from refractory organic compounds which may not exert an immediate demand, so the implications of high COD for oxygen demands within the reservoir are uncertain. BOD due to decomposition of vegetation declined to less than 12 percent of its initial value after 50 days. This suggests that oxygen demands from this source may be substantially lower after the first year, although oxygen demand due to flooded soils is expected to continue for several years.

Laboratory studies showed an initial 5-day BOD in the range of 5,740 to 20,500 mg/sq m for soil plus vegetation. Measurements at several stations in Richard B. Russell Lake for 1984 showed sustained high rates over several weeks. Thus, the total oxygen demand in the reservoir, as measured by areal depletion rates, easily exceeded the laboratory values for at least some of the stations (Table 7). Initially high values in the embayments may have resulted in part from high BOD of materials in the inflowing waters; this factor was not included in the laboratory studies.

Based on the above discussion, the amount of oxygen required to meet the discharge standard of 6 mg/l will be considerably higher than projected. Presented below are three methods to calculate oxygen requirements. The methods are arranged in order of increasing complexity, data needs, and accuracy.

Method one. The simplest approach treats the hypolimnion in its entirety in a single step; calculations can be made on a hand calculator. The following assumptions and suggested starting conditions are made for this approach: (1) The water volume to be oxygenated is approximately 144 million cu m (the hypolimnetic volume from the injection system to the dam). (2) The total daily flow is 12,500 cfs (30,400,000 cu m/day). (3) Oxygenation efficiency is assumed to be 75 percent. Efficiency is defined here to mean the proportion of injected oxygen that is either consumed to meet some demand in the forebay downstream of the injection system or released downstream. There is no consideration of absorption distribution. (4) Eighty percent of the total flow is

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assumed to occur from the hypolimnion (a typical value during the stratified period, using a hypolimnion top elevation of 134 m). Thus, any deficit, depletion, or oxygen surplus in the epilimnion (top 10-12 m) is not taken into consideration. (5) The maximum depletion rate for 1984 ranged from 0.1 to 0.5 mg/l per day (or 0.1 to 0.5 g/cu m per day); for the example below, 0.3 g/cu m per day will be used. (6) If most of the hypolimnion is anoxic, the oxygen deficit is at least 6 g/cu m plus an additional unknown debt. In this example 6 g/cu m will be used.

Calculations are made as follows:

(1) Oxygen consumed on a daily basis due to depletion in the forebay can be estimated by multiplying the depletion rate by the volume of the forebay hypolimnion. Oxygen required (OR) is calculated by dividing the amount of oxygen consumed by the assumed efficiency of the oxygenation system:

$$OR = \frac{HV \times OD}{E}$$

where:

HV = hypolimnetic volume (million cu m)

OD = oxygen depletion rate (g/cu m per day)

E = efficiency of oxygenation (percent/100)

Therefore,

$$OR = \frac{144 \times 0.3}{0.75}$$

$$= 58 \text{ metric tons per day}$$

(2) Oxygen consumed (OC) to satisfy the deficit/debt (from 6 g/cu m) in the approaching water can be estimated by multiplying the total daily flow from the hypolimnion by the amount of deficit/debt and dividing by the assumed efficiency of the oxygenation system:

$$OC = \frac{HF \times Q \times D}{E}$$

where:

HF = proportion of total flow from hypolimnion

Q = total daily flow (million cu m/day)

D = oxygen deficit/debt (g/cu m)

E = efficiency of oxygenation (percent/100)

Therefore,

$$OC = \frac{0.8 \times 30.4 \times 6}{0.75}$$

$$= 195 \text{ metric tons per day}$$

(3) The total oxygen required (TOR) due to depletion in the forebay and deficit in the inflowing water is obtained by summing the results above:

$$TOR = OR + OC$$

$$= 58 + 195$$

$$= 253 \text{ metric tons per day}$$

$$(= 278 \text{ short tons per day})$$

This high value reflects the high initial demand of the first year of filling. This value is expected to decrease in subsequent years as organic materials become oxidized.

For comparative purposes, the same calculation can be made using data from Clarks Hill Lake, as specified in the Design Memorandum (US Army Corps of Engineers, 1981). Using a depletion rate of 0.1 g/cu m per day and a maximum deficit of 3.8 g/cu m, the resulting oxygen requirement is 142 metric tons per day or about 156 short tons per day. This value compared well with 171 short tons per day as presented in US Army Corps of Engineers (1981).

Method two. A somewhat more complex approach divides the hypolimnion and epilimnion into layers and allows consideration of the selective withdrawal profile and epilimnetic dilution. The major features of this steady-state approach are: (1) The withdrawal profile at the dam face is calculated by SELECT (Bohan and Grace, 1973) based on the temperature profile at the dam and a total daily flow rate. (2) An upstream oxygen profile (i.e., above the injection system) is used for the deficit calculation. (3) A depletion rate is supplied to the model; this value is based on the change in successive oxygen profiles upstream of the injection system, for instance at Station 120. Since the depletion rate is appropriate only for the hypolimnion, the top elevation of the hypolimnion is also supplied. If there is no flow out of a particular layer of the hypolimnion (based on SELECT), that layer is not included in the calculation. (4) Oxygenation efficiency is assumed to be 75 percent.

Efficiency is defined as for the previous method. (5) Consumption due to depletion is calculated for each 1-m layer in a manner similar to the previous method; layer volume for the forebay hypolimnion is multiplied by the rate of depletion, which is assumed constant for all layers. Oxygen consumption is added for all hypolimnion layers from which there is any withdrawal at the dam. (6) Consumption due to deficit includes the entire water column, so that epilimnetic dilution, if any, can be accounted for. Flow from a layer is multiplied by the deficit from or surplus over 6 g/cu m to yield a mass of oxygen required or in excess for that layer. Values are summed to yield a mass required for the entire water column. (7) Total oxygen required is calculated by adding consumption due to deficit and depletion and dividing by the efficiency.

The steady-state approach, as presently conceived, has the following deficiencies: (1) There is no account of the oxygen debt once the hypolimnion becomes anoxic; there could be a large demand which is not considered. (2) There is no means of accounting for transient behavior; outflow is assumed to be continuous when, in fact, high flow may occur during a short period of the day, with little or no flow for the rest of the day.

Results using the steady-state approach and 1984 data are shown in Figures 42-44; simulations at each of three flow rates were made for depletion rates ranging from 0.1 to 0.4 g/cu m per day. All data profiles from April through December were used to obtain the results shown. Since the injection system was not in operation for 1984, oxygen profiles from Station 60 were used to calculate the deficit from 6 g/cu m. Note that at the highest flow rate, the oxygen requirement approached 350 short tons per day, without considering the debt. At the highest flow rate, the oxygen required to satisfy the deficit is a much greater proportion of the total oxygen requirements than the amount needed to offset depletion in the forebay below the injection system. Also note that the relative contribution of the depletion rate increases as the flow rate decreases.

Nomographs showing oxygen requirements at three flow rates, given an overall deficit and depletion rate, are shown in Figures 45-47. These figures were derived from results of the steady-state model based on a typical selective withdrawal profile for the stratified period. The depletion rate was applied to the hypolimnion (elevation 135 m and below); the deficit represents an overall value for the water column, including the epilimnion. This value
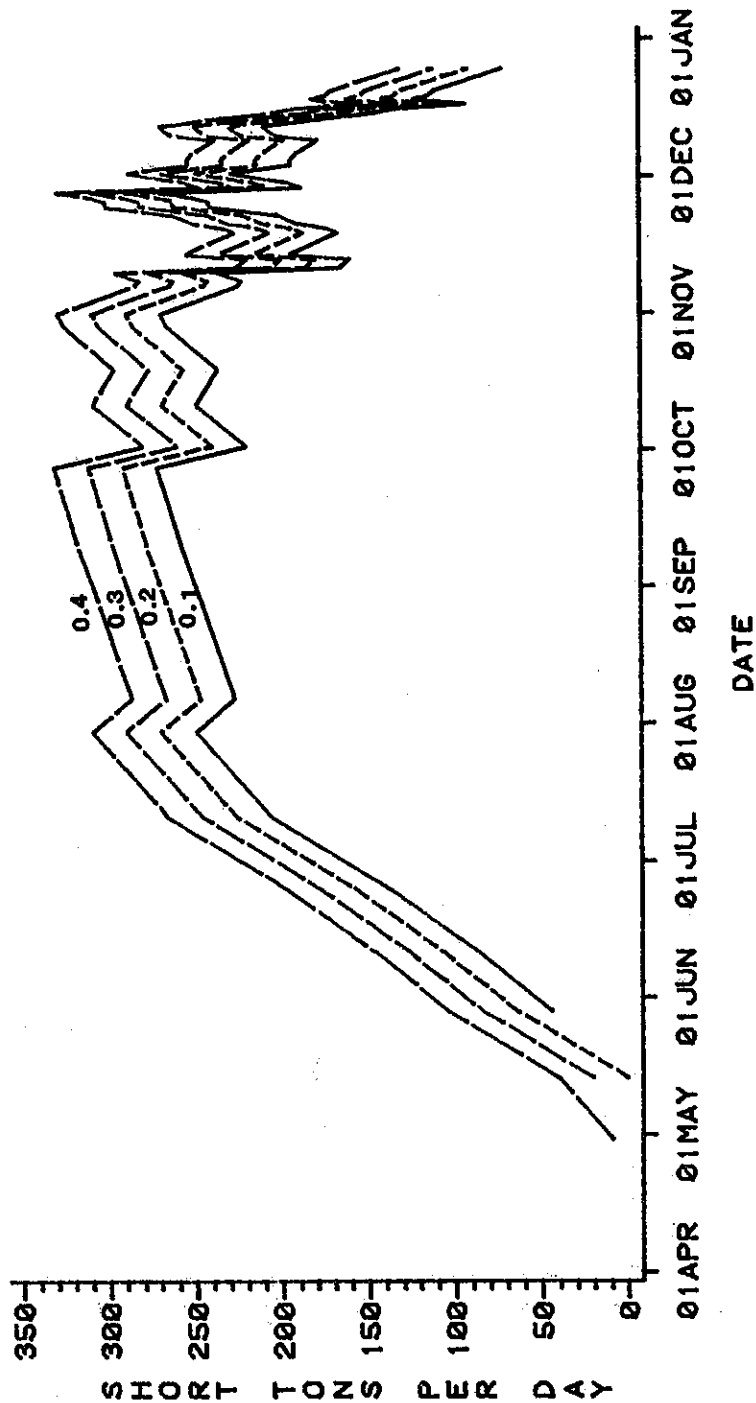
Figure 42. Dissolved oxygen requirements for Richard B. Russell Lake based on
a total discharge rate of 12,500 cfs and dissolved oxygen depletion rates of
0.1, 0.2, 0.3, and 0.4 mg/l per day. Requirements assume 75-percent oxygen-
ation efficiency and do not include an oxygen debt
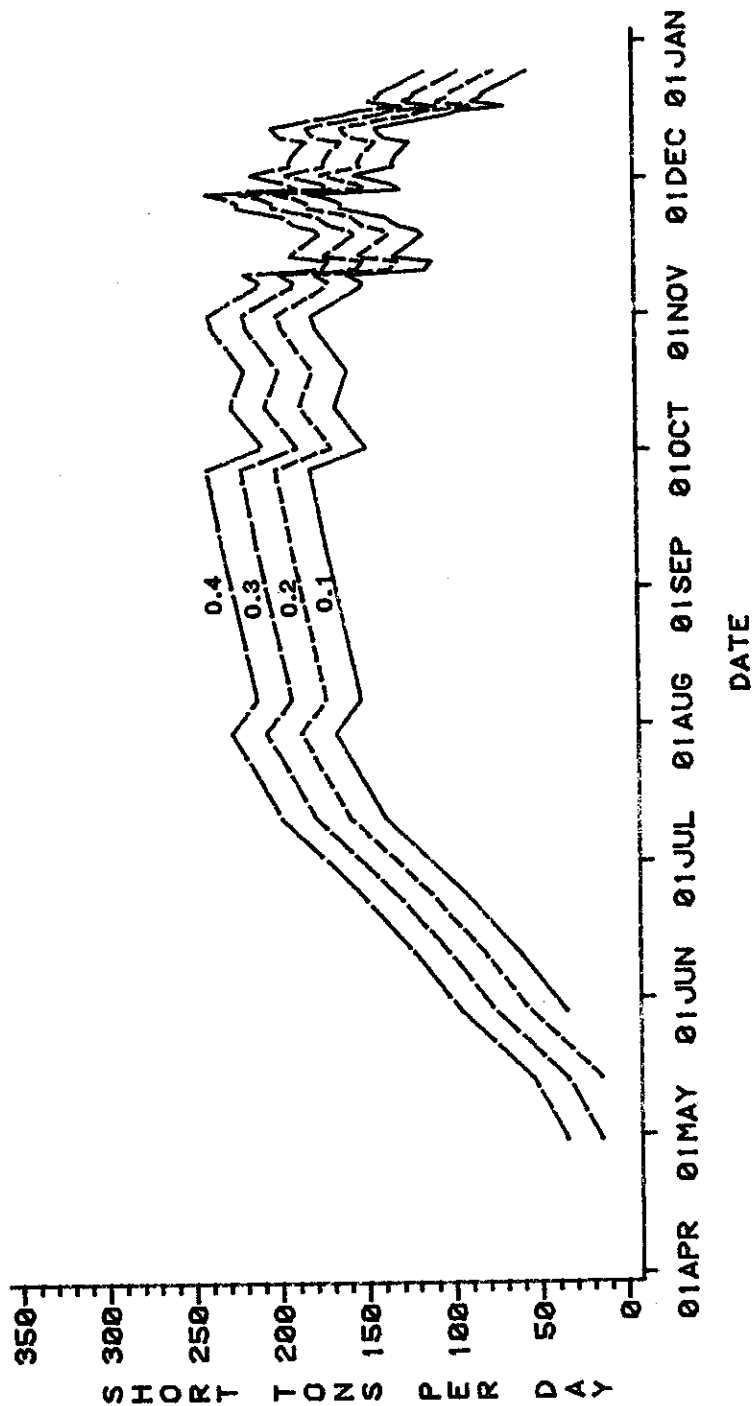
Figure 43. Dissolved oxygen requirements for Richard B. Russell Lake based on a total discharge rate of 8,000 cfs and dissolved oxygen depletion rates of 0.1, 0.2, 0.3, and 0.4 mg/l per day. Requirements assume 75-percent oxygenation efficiency and do not include an oxygen debt
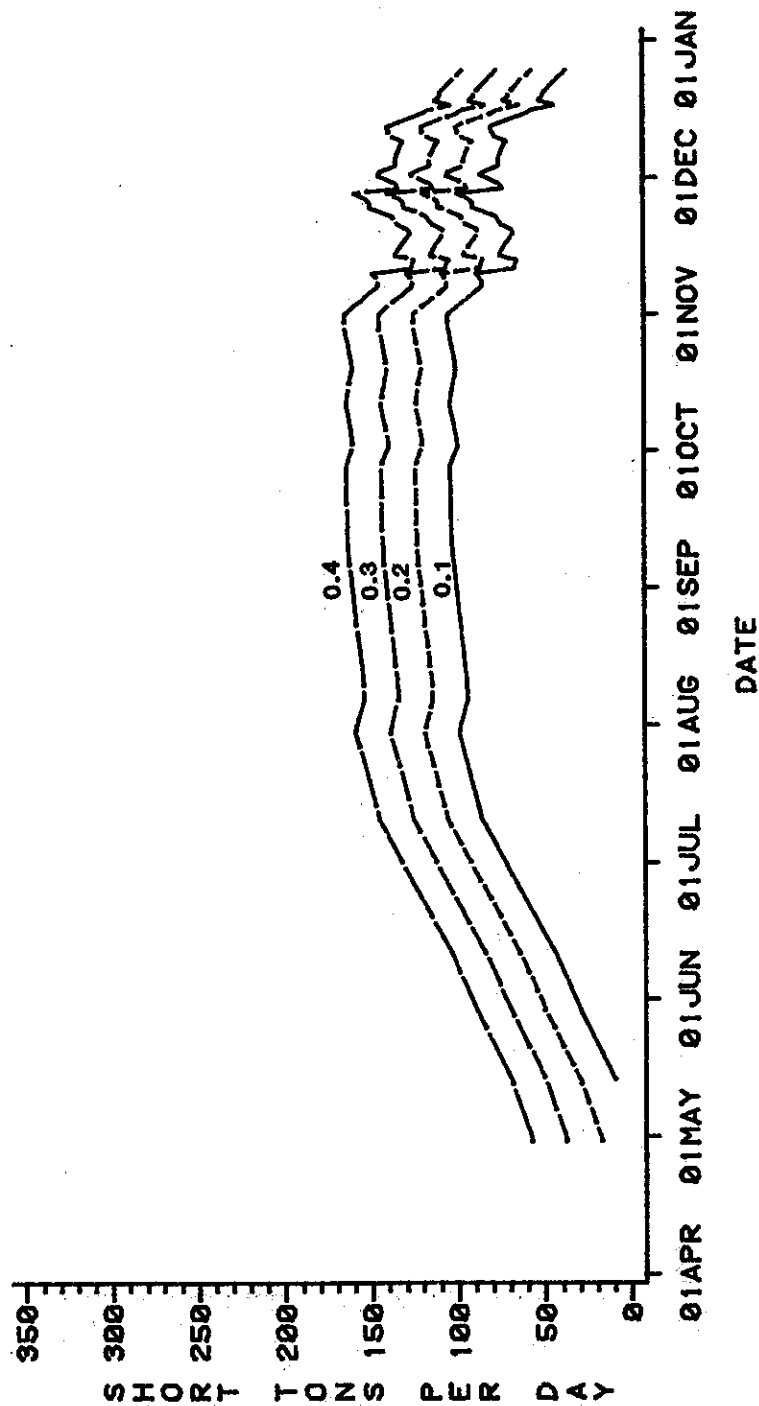
Figure 44. Dissolved oxygen requirements for Richard B. Russell Lake based on a total discharge rate of 4,000 cfs and dissolved oxygen depletion rates of 0.1, 0.2, 0.3, and 0.4 mg/1 per day. Requirements assume 75-percent oxygenation efficiency and do not include an oxygen debt
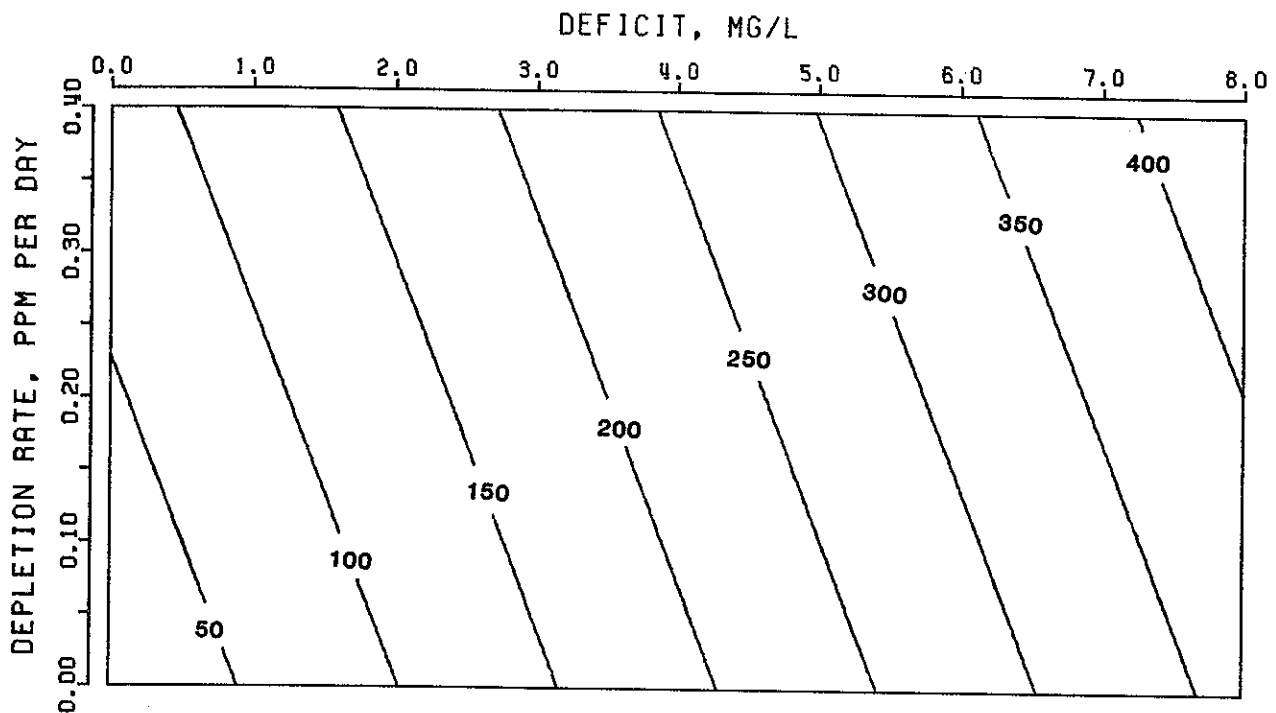
Figure 45. Nomograph of oxygen requirements (short
tons per day) in Richard B. Russell Lake based on a
total discharge rate of 12,500 cfs
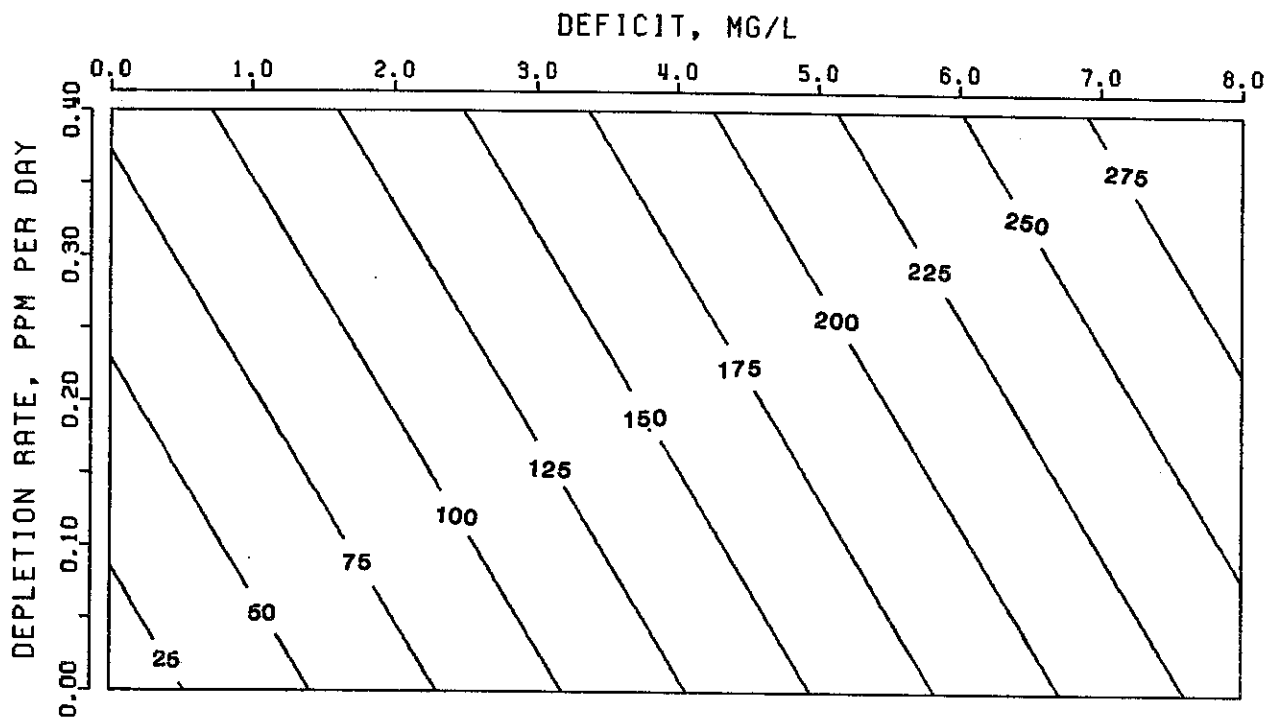
Figure 46. Nomograph of oxygen requirements (short
tons per day) in Richard B. Russell Lake based on a
total discharge rate of 8,000 cfs

DEFICIT, MG/L
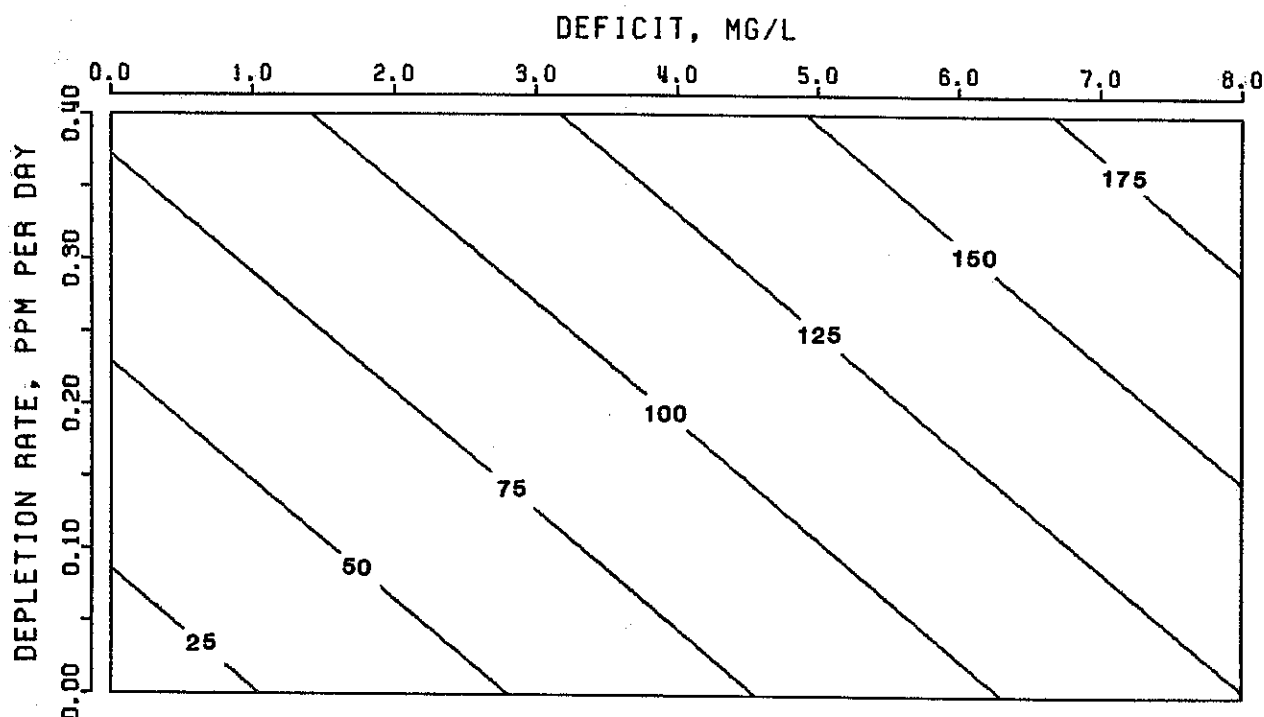
Figure 47. Nomograph of oxygen requirements (short
tons per day) in Richard B. Russell Lake based on a
total discharge rate of 4,000 cfs

could include a debt estimate if available. These figures can be used to
estimate oxygen requirements for various flow regimes, given some projected
depletion rate and overall deficit value. The overall deficit (not includ-
ing any debt) can be obtained from a flow-weighted average of an upstream
oxygen profile. Values for the debt could be added as analytical methods
are developed. Proportions of flow from each layer can be obtained using
SELECT.

Method Three. Estimates of oxygen requirements can potentially be made
with greater precision and accuracy using a two-dimensional hydrodynamic and
water quality model. Such a model will allow consideration of nonsteady-state
conditions and longitudinal and vertical differences. The model is based on
CE-QUAL-W2 (derived from LARM; Environmental Laboratory, 1985). Its use for
this particular application will be considerably simplified and empirically
based to allow its use in a more rapid predictive mode. The basic concepts
of the model are shown in Figure 48. The purpose of the model is to predict,
based on the criteria listed below, the downstream oxygen profile near the
dam as well as the outflow oxygen concentration.
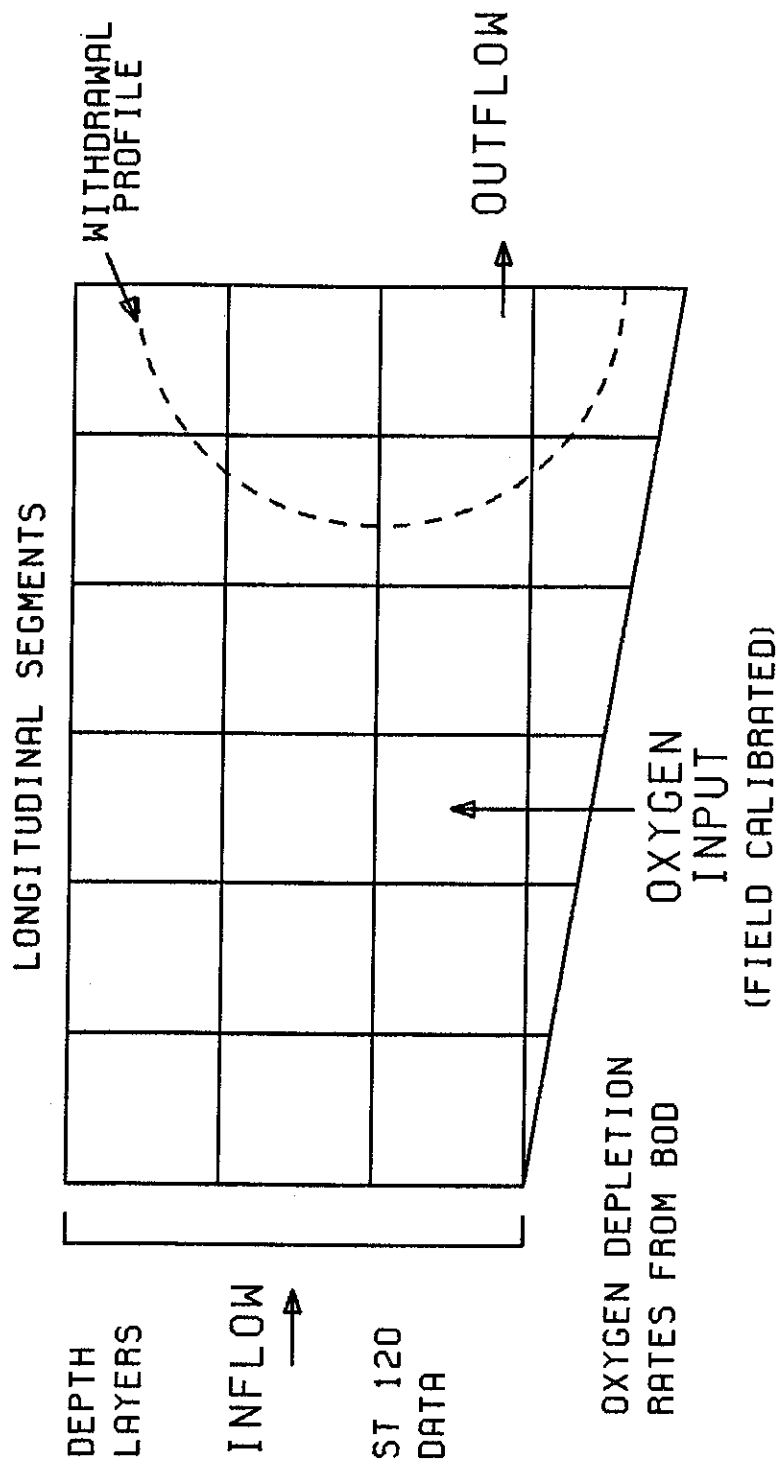
Figure 48. Diagram of a two-dimensional hydrodynamic and water quality model for Richard B. Russell Lake

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Development of the two-dimensional model is centered around the following points: (1) The area to be modeled will be restricted to the forebay of the reservoir, approximately the lower 5 km. (2) In-situ measurements of oxygen and temperature profiles upstream of the injection system will be used for initial conditions. The initial conditions will therefore include the oxygen deficit in the model calculations (but not the oxygen debt until a method is devised to include it). (3) Oxygen depletion can initially be obtained from changes in the in-situ oxygen profiles in the same manner as for the steady-state approach. (4) Oxygen input from the injection system will initially be estimated using published results from Clarks Hill Lake. These estimates will need to be improved as field measurements become available. (5) The withdrawal zone will be calibrated using SELECT. (6) Data for 1-week periods will be used as input for the model. Meteorological data will not be used since heat exchange will not be considered. Heat exchange should not be important over the short time period between updates using measured profiles.

The model as described initially above would probably not prove to be more accurate than the steady-state approach unless some estimate of oxygen debt and depletion rates can be obtained when anoxic conditions have been reached. Measurements of the rate of oxygen utilization of a water sample would give a more precise estimate of the actual rate of depletion, especially under anoxic conditions. In-situ oxygen demand in the field would also be required.

Oxygen requirements will remain higher in Richard B. Russell Lake than projected based on Clarks Hill Lake for at least 3 to 5 years, as suggested by: (1) demands during the first year following filling, (2) laboratory studies of vegetation decomposition and demand due to flooded soils in the reservoir, and (3) previous experience with newly filled reservoirs. The total amount of oxygen needed should gradually decline and be spread out over a gradually diminishing proportion of the stratified period. A more precise estimate of future requirements is not possible at this time because of uncertainties in long-term decomposition rates, withdrawal operation schedules, and efficiency of the injection system. A better estimate should be possible after the second year of reservoir operation when more information will be available on in-situ demands, as the performance of the injection system is evaluated, and as the predictive methods outlined above are improved.

Seasonal patterns in the thermal regime of Clarks Hill Lake were char-
acteristic of a monomictic lake and typical for reservoirs in the southeast-
ern United States. The lake experienced isothermal and well-mixed conditions
in late fall and winter, and pronounced and stable stratification during the
summer months. Changes in the thermal structure at Station 20 were indica-
tive of seasonal and vertical changes throughout the lake (Figure 49). Near-
isothermal conditions were evident from January until April, with the water
column temperature gradually warming during this period from a low of
8 deg. C to approximately 12 deg. C. Seasonally warmer air temperatures in
early spring led to surface warming trends in May. On 31 May a distinct ther-
mocline was evident between the 10- and 14-m depth, and temperatures ranged
from approximately 23 deg. C at the surface to 13 deg. C at the bottom.

The period of well-defined thermal stratification lasted from June until
late September. During this period, temporal changes in the thermal structure
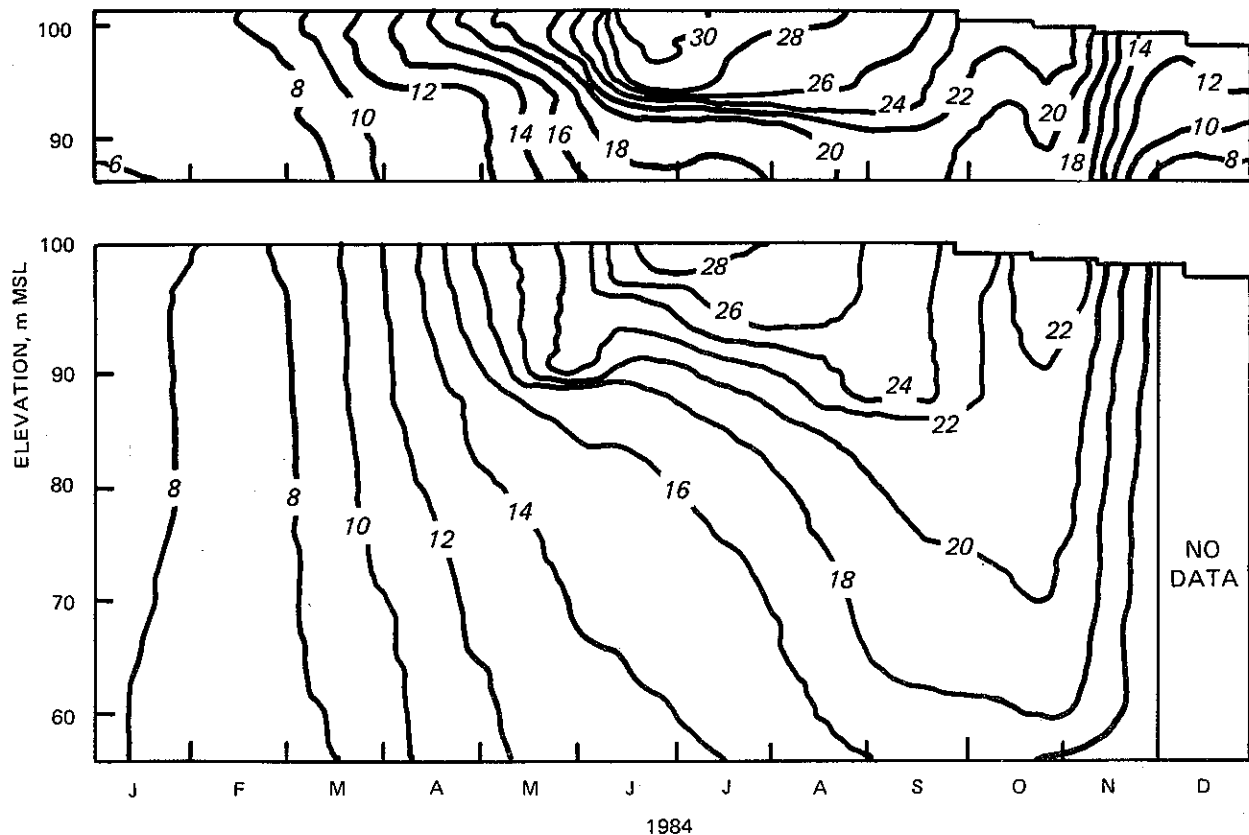
Figure 49. Temporal and vertical patterns in temperature (deg. C) at
Station 40 (upper panel) and Station 20 (lower panel) during 1984

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reflected changes in weather conditions throughout the summer months (Figure 49). Continued surface warming in late spring resulted in well-defined stratification patterns in June and July with temperature maxima near 30 deg. C in the epilimnion. In early July, the epilimnion was apparent from the surface to the 3-m depth, and the metalimnetic region occurred from 4 to 10 m. From late-July through September, seasonally cooler air temperatures and wind mixing led to surface cooling and epilimnetic expansion. By 11 September epilimnetic temperatures had decreased to approximately 25 deg. C and the mixed zone had increased in depth to 13 m. Hypolimnetic temperatures exhibited gradual increases throughout the summer period with bottom temperatures ranging from 14.1 deg. C in July to 16.2 deg. C in September.

Thermal destratification at Station 20 began in late September, and isothermal conditions were observed by mid-November. Temperature ranged from 22 deg. C at the surface to 16.1 deg. C at the bottom in mid-October. No clearly defined thermocline was observed (Figure 49). On 13 November, a major percentage of the water column exhibited isothermal conditions. Data collected on 15 January indicated complete water column mixing with a uniform temperature of 10.8 deg. C.

Temporal changes in thermal structure were slightly different at Station 40 due to influences of a shallow basin morphometry and to coolwater releases from Richard B. Russell Dam. Surface warming was more rapid here, and stratified conditions were evident in April (Figure 49). A well-established thermal structure existed in June, and temperatures ranged from over 30 deg. C at the surface to 17 deg. C at the bottom. Throughout the stratified period, which at this station lasted from May until approximately October, the depth of the epilimnetic mixing zone remained fairly constant at 6 to 7 m. During this period, hypolimnetic temperatures increased more rapidly than temperature changes occurring at Station 20. Temperatures increased from 12 deg. C in May to over 20 deg. C in late September. Near-isothermal conditions were established in early November, a month earlier than at Station 20.

Seasonal and spatial patterns in dissolved oxygen concentration in Clarks Hill Lake were dissimilar to those observed in Richard B. Russell Lake due to differences in basin age and morphometry. In general, significant hypolimnetic dissolved oxygen depletion was restricted to the near-dam and mid-reservoir regions of the pool. Hypolimnetic anoxia occurred at Station 20 for only a short period of time late in the stratified period. In addition,

the release of water from Russell Dam tainter gates provided a continuing influx of well-oxygenated water to the headwater region of Clarks Hill Lake.

Seasonal patterns in dissolved oxygen for Station 20 are presented in Figure 50. During isothermal conditions, oxygen concentrations were near saturation (i.e., >10 mg/l) and uniform throughout the water column. Coincident with the onset of stratified conditions was a gradual decline in hypolimnetic dissolved oxygen concentrations and the occurrence of a dissolved oxygen minimum in the metalimnetic region. Dissolved oxygen began declining at the 10- to 20-m depth zone in June, and levels less than 2.0 mg/l were detected in late-July to August. Thereafter, convective mixing and epilimnetic expansion resulted in the introduction of dissolved oxygen to these depths. In the hypolimnion, dissolved oxygen exhibited a gradual decline from >6.0 mg/l in May to <1.0 mg/l in August. Hypolimnetic anoxia was evident at depths below 25 m from September until mid-November. Mixing and turnover resulted in increased and nearly uniform concentrations throughout the water column. Concentrations during this period exceeded 8.0 mg/l.
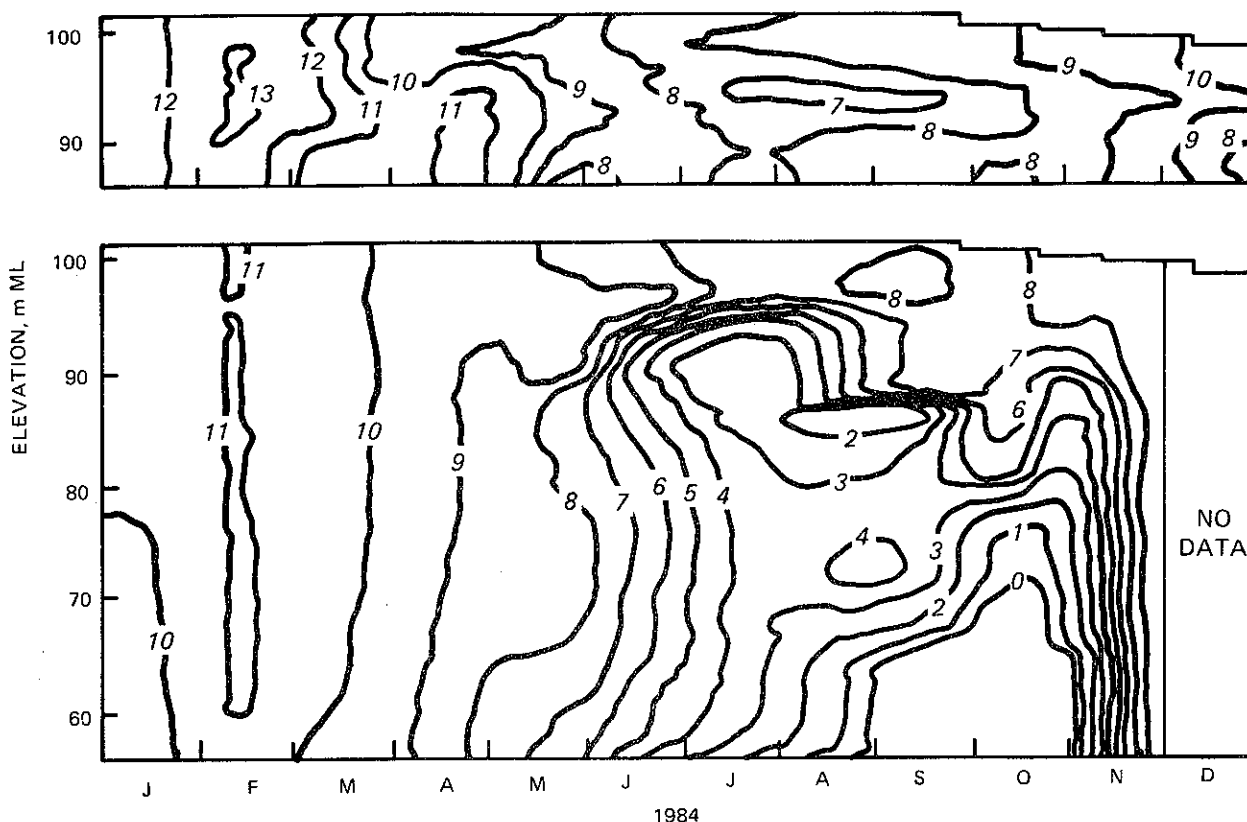
Figure 50. Temporal and vertical patterns in dissolved oxygen (mg O_2/l) at Station 40 (upper panel) and Station 20 (lower panel) during 1984

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Station 40 exhibited a different seasonal pattern in the distribution
of dissolved oxygen (Figure 50). Concentrations remained above 6.0 mg/l
throughout the water column during the stratified period, and releases from
Richard B. Russell Dam appeared to have a marked impact on these patterns.
As will be discussed in more detail later, inflows from tainter gate releases
from Richard B. Russell Dam entered Clarks Hill Lake as an interflow upstream
from Station 40. Passage of release water down the dam face and over the
flip buckets provided sufficient reaeration to markedly increase dissolved
oxygen concentrations. During the latter part of the stratified period,
slight concentration increases were detected from the 8-m depth to the
bottom at Station 40, suggestive of the movement of these inflows through
the headwater region as a density current. Station 40 exhibited a similar
trend of high dissolved oxygen concentrations during the late-winter mixed
period.

Longitudinal patterns in temperature, dissolved oxygen, and specific
conductance observed on 1 May (Figure 51), 23 July (Figure 52), and 12 October
(Figure 53) were exemplary of lakewide trends. In spring, weak thermal strat-
ification was observed from the dam to near Station 40 (Figure 51). Strati-
fied conditions were more pronounced in the shallower headwater area as cool
releases from Richard B. Russell Dam plunged upstream of Station 40 and con-
tinued downstream as a density current. The occurrence of these density in-
flows was further suggested by high dissolved oxygen concentrations at bottom
depths in the headwater region and by lower values of specific conductance.
Dissolved oxygen concentration exceeded 11 mg/l in the hypolimnion at Station
40, and a zone of water exhibiting concentrations greater than 10 mg/l was de-
tected at bottom depths near Station 30.

In summer, thermally stratified conditions were well established along
the length of Clarks Hill Lake, and releases from Richard B. Russell Dam con-
tinued to progress through upstream portions of Clarks Hill Lake as an inter-
flowing density current. On 23 July (Figure 52), an epilimnion was evident
from the lake surface to the 6-m depth along much of the lake's length.
Dissolved oxygen patterns indicated a zone of high dissolved oxygen concen-
tration from Station 40 to Station 30 in the area of density interflow.
This zone was located below the thermocline. Specific conductance values
were lower in this area and reflective of values observed in the Richard B.
Russell Dam tailwater area. In the deeper downstream areas of Clarks Hill
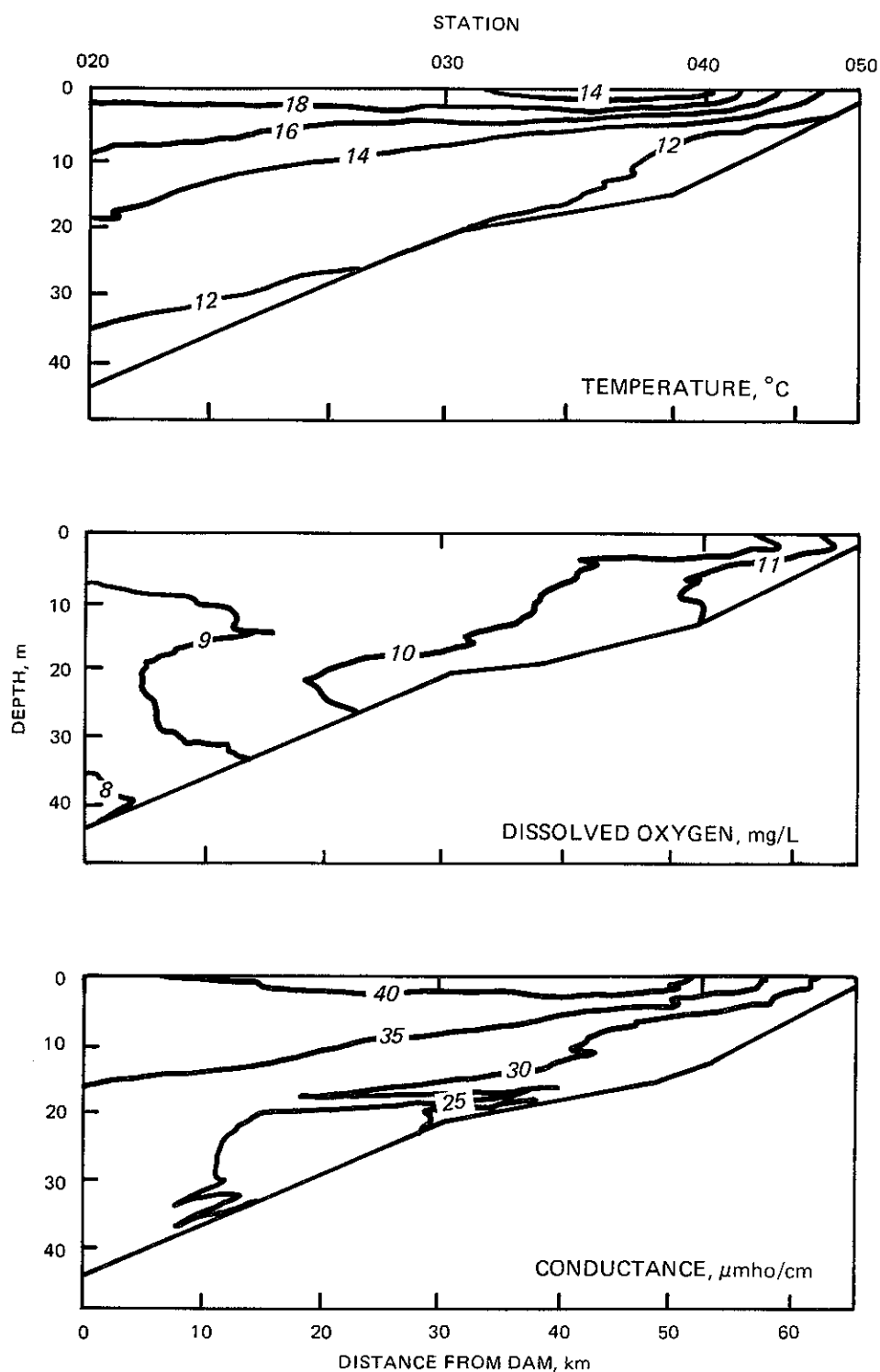
Figure 51. Vertical and longitudinal patterns in tempera-
ture, dissolved oxygen, and specific conductance for the
main basin of Clarks Hill Lake on 1 May 1984
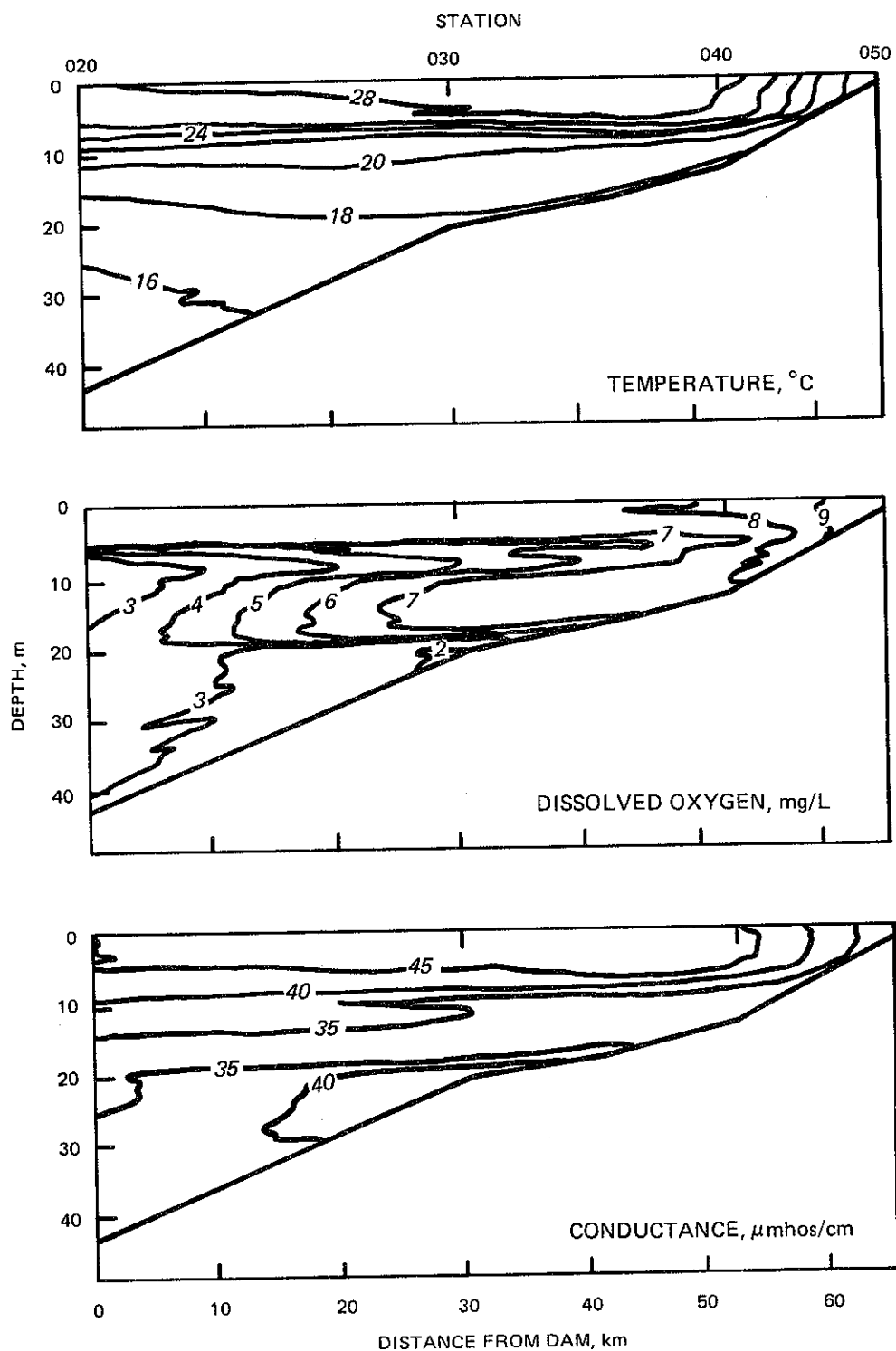
Figure 52. Vertical and longitudinal patterns in temperature, dissolved oxygen, and specific conductance for the main basin of Clarks Hill Lake on 23 July 1984
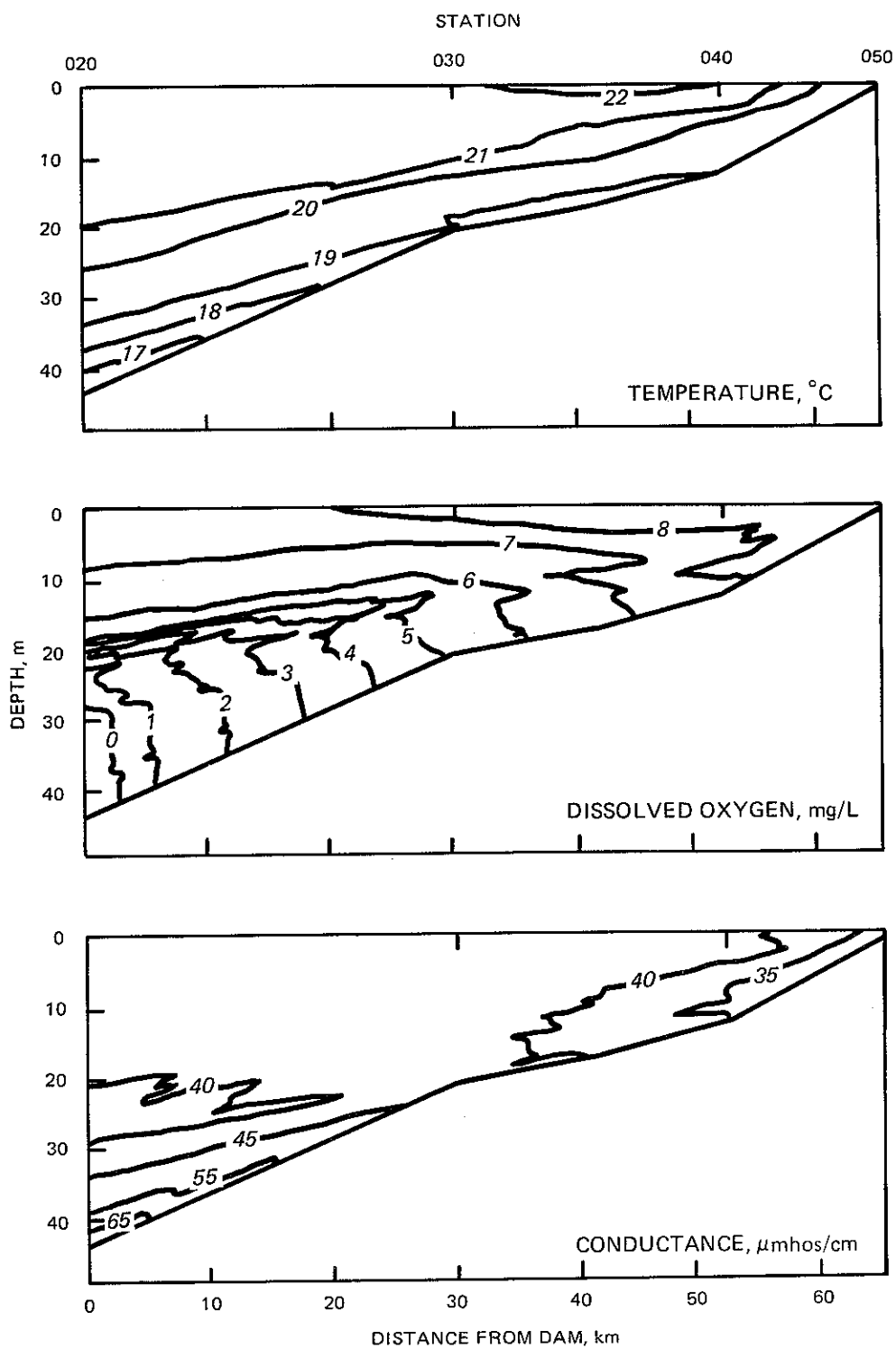
Figure 53. Vertical and longitudinal patterns in tempera-
ture, dissolved oxygen, and specific conductance for the
main basin of Clarks Hill Lake on 12 October 1984

Lake, hypolimnetic dissolved oxygen depletion was evident with dissolved oxygen levels near 3.0 mg/l.

In fall, longitudinal patterns in in-situ characteristics were observed during destratification. Surface temperatures on 12 October (Figure 53) had decreased to 22 deg. C along the lake's length, and mixing was evident to a depth of 20 m in the forebay area. However, the headwater region exhibited moderate temperature gradients near the lake's surface, possibly due to intermittent surface heating. Evident from dissolved oxygen and specific conductance values was the continued influence of density inflows on the upper reaches of Clarks Hill Lake. Dissolved oxygen concentrations remained high in the area of interflow. However, hypolimnetic anoxia continued to be observed in the forebay region, and reduced concentrations were observed toward Station 30.

Areal oxygen depletion rates (mg O_2/sq m/day) for various stations in Clarks Hill Lake are presented in Table 8. In general, areal rates calculated for the combined metalimnion and hypolimnion were lower in Clarks Hill Lake than in Richard B. Russell Lake. Values ranged from 1,272 mg O_2/sq m/day at Station 40 to 1,458 mg O_2/sq m/day at Station 30, suggesting little longitudinal variation in the reservoir. Bottom waters at Station 30 exhibited the highest areal hypolimnetic rate. This rate, 3,185 mg O_2/sq m/day, was comparable to values observed at Stations 130 and 160 in Richard B. Russell Lake.

Seasonal and spatial patterns in chemical conditions observed in Clarks Hill Lake were less distinct than those observed in Richard B. Russell Lake. For instance, total and dissolved organic carbon, total iron, and ammonia nitrogen exhibited much lower hypolimnetic concentrations in the forebay area of Clarks Hill Lake (i.e., Station 20) than in Richard B. Russell Lake. Exceptions were total manganese and total and dissolved nitrogen concentrations, which were comparable among the two reservoirs.

Organic carbon displayed few seasonal and spatial fluctuations during the study period, as indicated from data collected at Stations 20 and 40 (Figure 54). Total organic carbon concentrations were high during late winter and early spring (e.g., during periods of storm events and high flow) and lower in summer. In summer, total organic carbon concentrations were generally higher in the epilimnion than in the hypolimnion, due possibly to increased algal productivity during this period. Slightly elevated concentrations were detected at Station 40 in late winter; however, epilimnetic and hypolimnetic

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Figure 54. Temporal and vertical patterns in total organic carbon (mg C/l) at Station 40 (upper panel) and at Station 20 (lower panel) during 1984

concentrations were similar in summer. Dissolved organic carbon concentrations followed a similar spatial and seasonal trend.

Complex longitudinal and vertical patterns in nitrogen concentrations were observed in Clarks Hill Lake during 1984 (Figure 55). Overall, concentrations were lowest during winter and highest in the summer. In the epilimnion at both stations, total nitrogen exhibited small seasonal concentration increases from April to September, and between-station differences were minimal. Concentrations ranged from 0.35 to 0.64 mg N/l at Station 20 and from 0.55 to 0.63 mg N/l at Station 40. In the hypolimnion, concentrations were elevated during summer stratification, and only moderate between-station differences were evident. Station 20 exhibited elevated concentrations (i.e., >0.8 mg N/l) at bottom depths from May until September. Dissolved nitrogen and ammonia nitrogen displayed similar hypolimnetic concentration increases during stratification. For example, ammonia-nitrogen increased at this depth

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Figure 55. Temporal and vertical patterns in total nitrogen (mg N/l) at
Station 40 (upper panel) and at Station 20 (lower panel) during 1984

from an undetectable level in April to a maximum level of 0.25 mg N/l at Station 20 on 11 September. At Station 40, the total nitrogen concentration at the bottom depth increased to near 0.70 mg N/l in August.

Total phosphorus concentrations were higher throughout the water column at Stations 20 and 40 in late winter and lower during stratification (Figure 56). Highest total phosphorus concentrations were observed at Station 40, particularly in the late winter. Reasons for these patterns are unclear; however, they may have been related, in part, to influences of releases from Richard B. Russell Dam. For instance, bottom depth concentrations at Station 40 were 0.044 mg P/l in April, and thus similar to the concentration observed at Richard B. Russell Dam (i.e., at Station 50), which averaged 0.042 mg P/l during this period.

Moderate increases in hypolimnetic total phosphorus concentrations during summer stratification contrasted markedly with increases observed at

Figure 56. Temporal and vertical patterns in total phosphorus (mg P/l) at Station 40 (upper panel) and at Station 20 (lower panel) during 1984

Richard B. Russell Lake. At Station 40, hypolimnetic concentrations were low and strongly influenced by discharge waters from Richard B. Russell Dam. Despite the occurrence of hypolimnetic anoxia at Station 20 in September, total phosphorus concentration remained low. Generally, phosphorus release from sediments is often associated with a reduction of iron. Therefore, concomitant increases in soluble iron and phosphorus are often observed in hypolimnia of lakes and reservoirs during anoxia. However, this mechanism of release may not play an important role in phosphorus cycling in Clarks Hill Lake since little soluble iron was detected at Station 20 during anoxia.

Total iron concentrations exhibited pronounced differences between Station 20 and Station 40 during the spring and fall mixing season, and similar concentrations during the stratified period (Figure 57). Station 40 exhibited elevated concentrations of 2-4 mg Fe/l total iron in the bottom waters from January until March and surface concentrations near 3 mg Fe/l in April.

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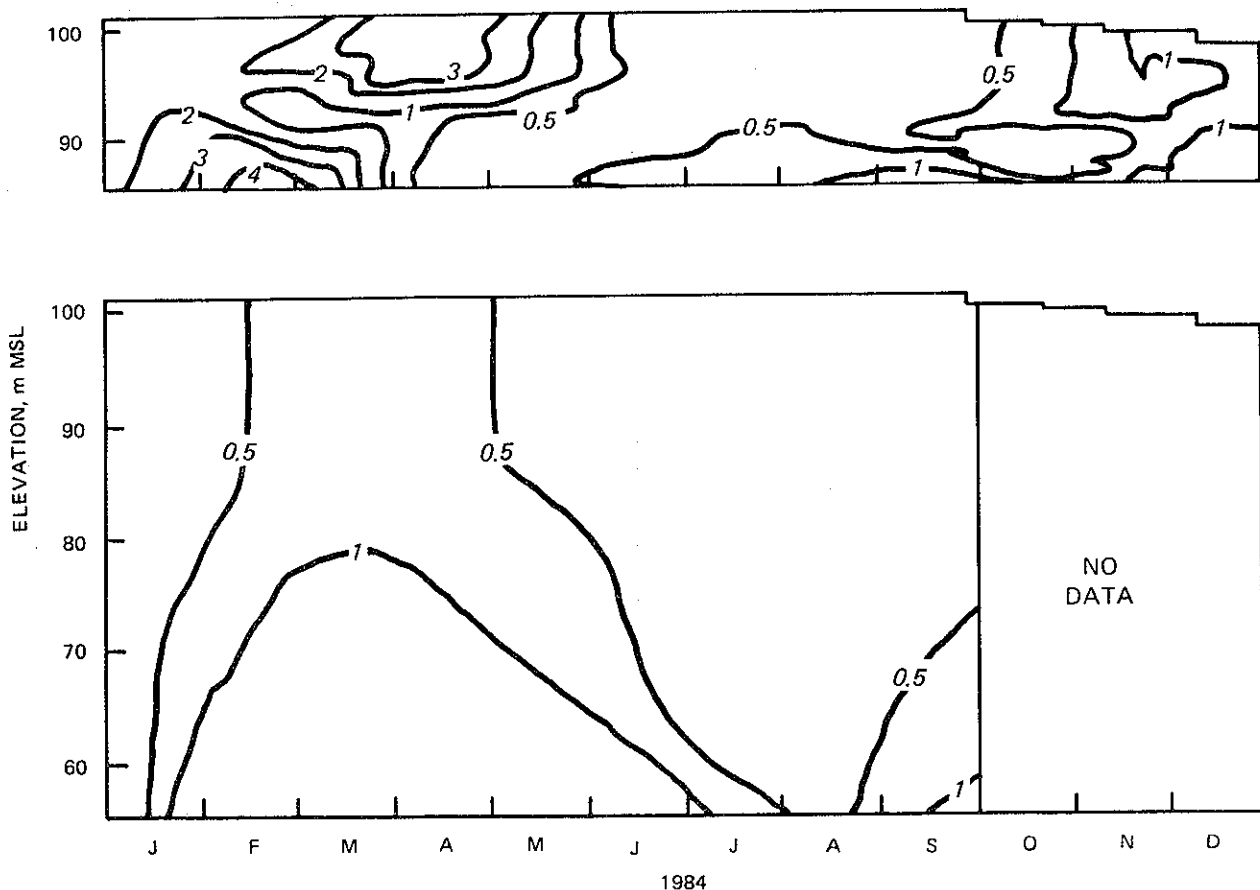
Figure 57. Temporal and vertical patterns in total iron (mg Fe/l) at Station 40 (upper panel) and at Station 20 (lower panel) during 1984

Concentration throughout the water column declined to 0.5 mg Fe/l during the stratified period, then increased slightly to 1.0 mg Fe/l after destratification. Concentrations of total iron at Station 20, however, were seasonally lower and did not exhibit marked fluctuations. These longitudinal differences may have been related to the influences of releases from Richard B. Russell Dam. In addition, hypolimnetic total iron concentrations did not increase in the forebay area during anoxia.

Total manganese concentrations exhibited a longitudinal pattern that was dissimilar to that observed for total iron. Unlike iron, total manganese concentrations exhibited a large increase in the hypolimnion of the forebay area during the latter period of thermal stratification and anoxia. Furthermore, between-station differences were apparent for manganese. While manganese concentrations were always low at Station 40, levels at Station 20 were below 1.0 mg Mn/l from March until July, then increased to 3.9 mg Mn/l in September

(Figure 58). This increase was apparently related to the occurrence of anoxia and the establishment of an environment favorable for reduction of manganese hydroxides. Over 90 percent of the total manganese was in the soluble form, and concentrations were comparable to those observed in the forebay region of Richard B. Russell Lake.

These differing responses between total manganese and total iron in the forebay area may be a result of the redox potential in the hypolimnion. Manganese becomes soluble at a higher redox potential than iron and, therefore, is often released from the sediment before iron. In contrast, total manganese levels at Station 40 ranged from undetectable concentrations to 0.3 mg Mn/l in the hypolimnion, and reflected the low levels of manganese in the release waters from Russell Lake. Levels at the epilimnetic depth were low throughout the study period and frequently undetectable.
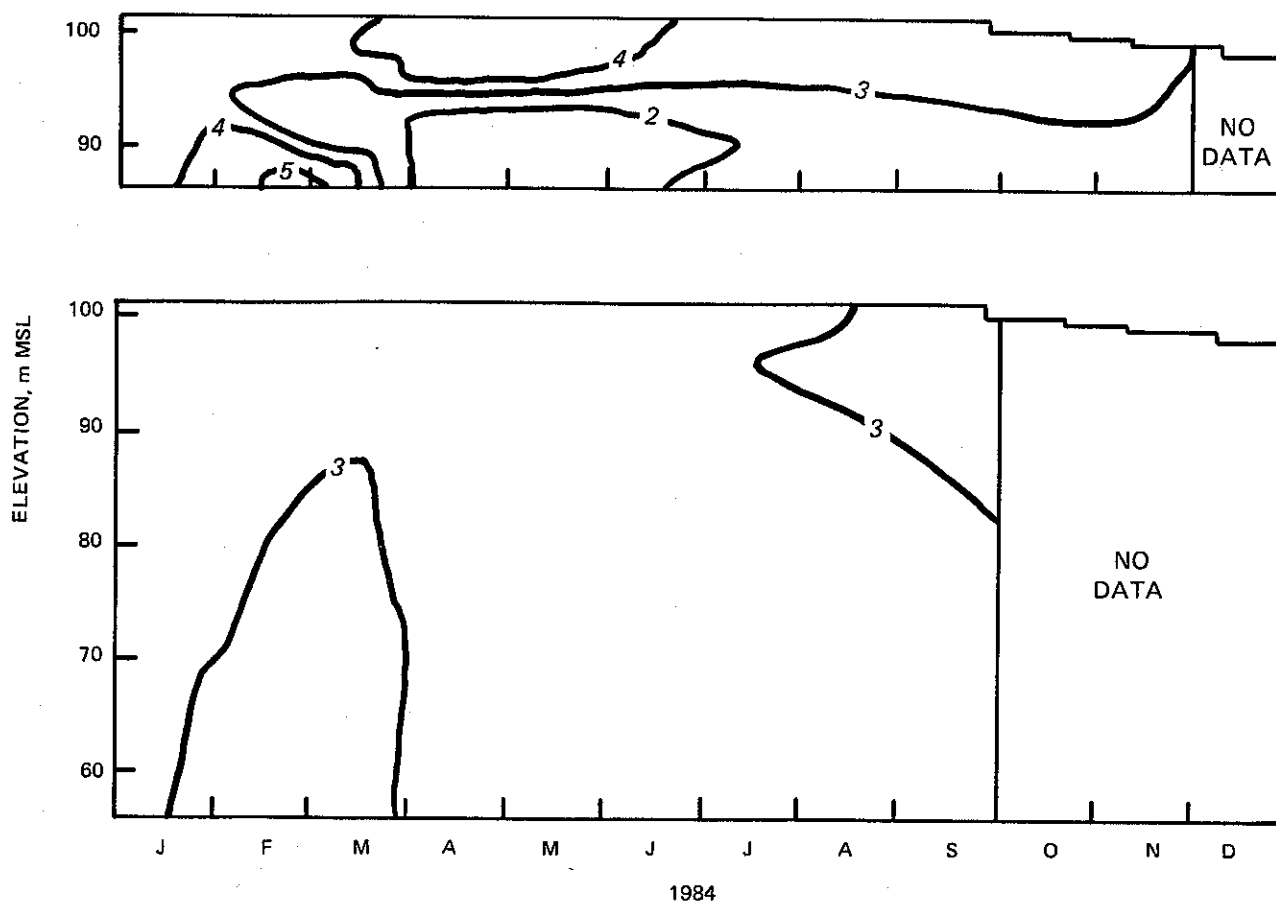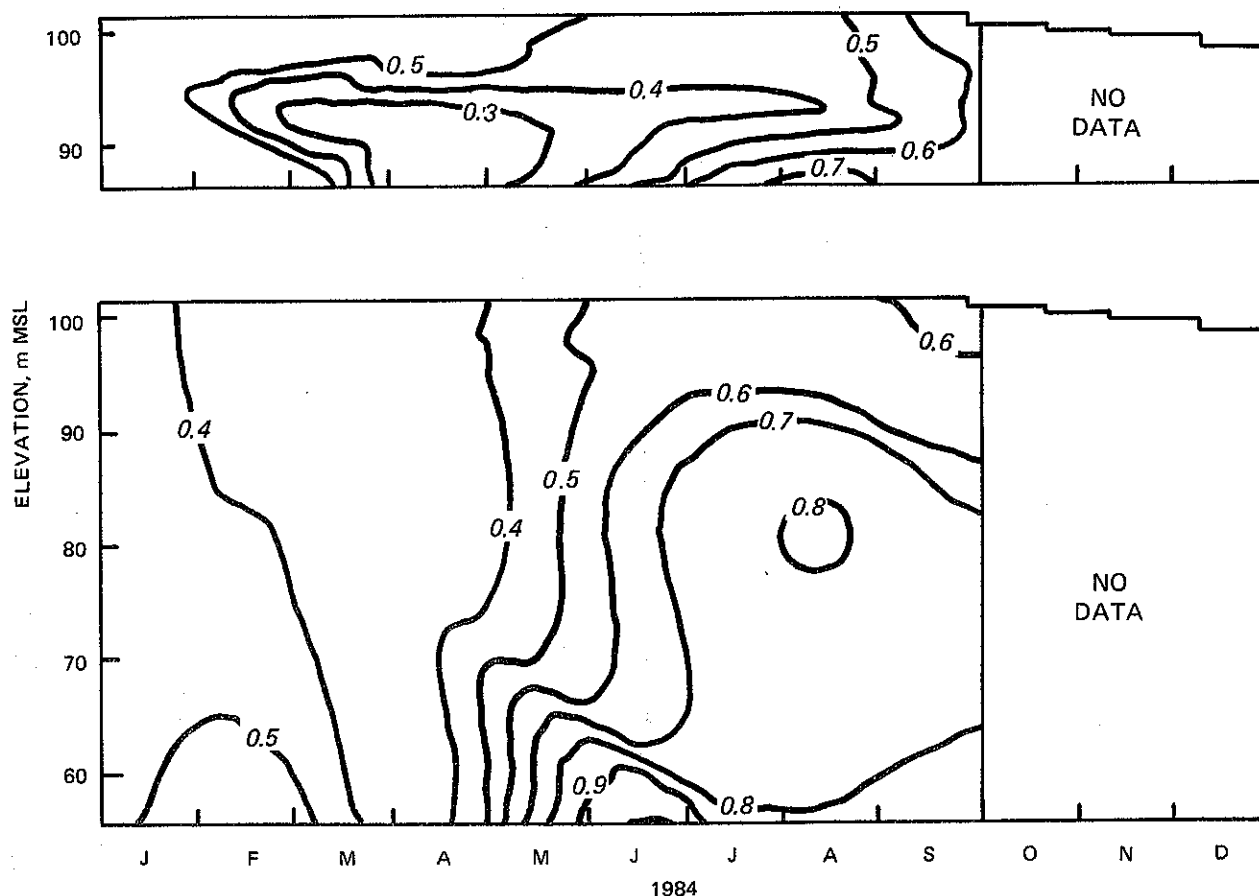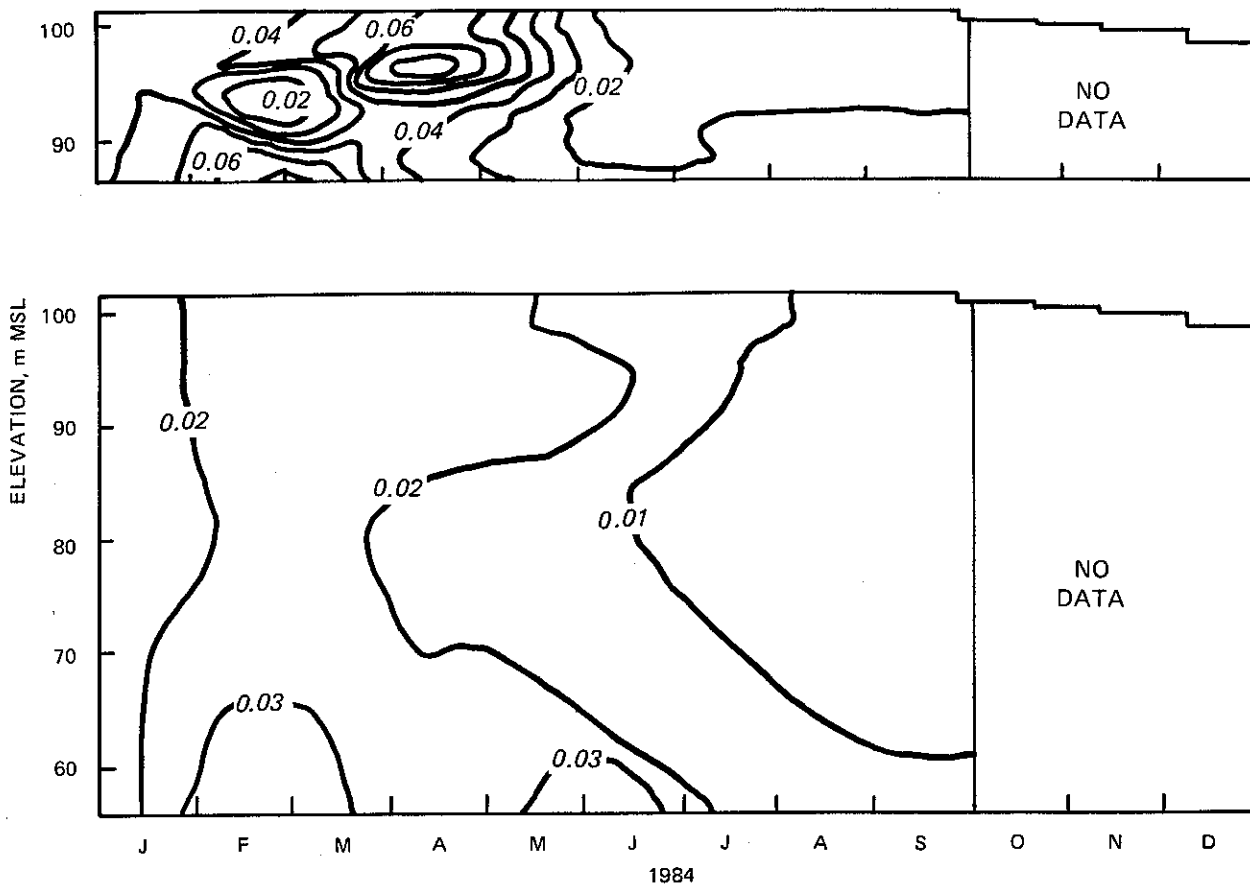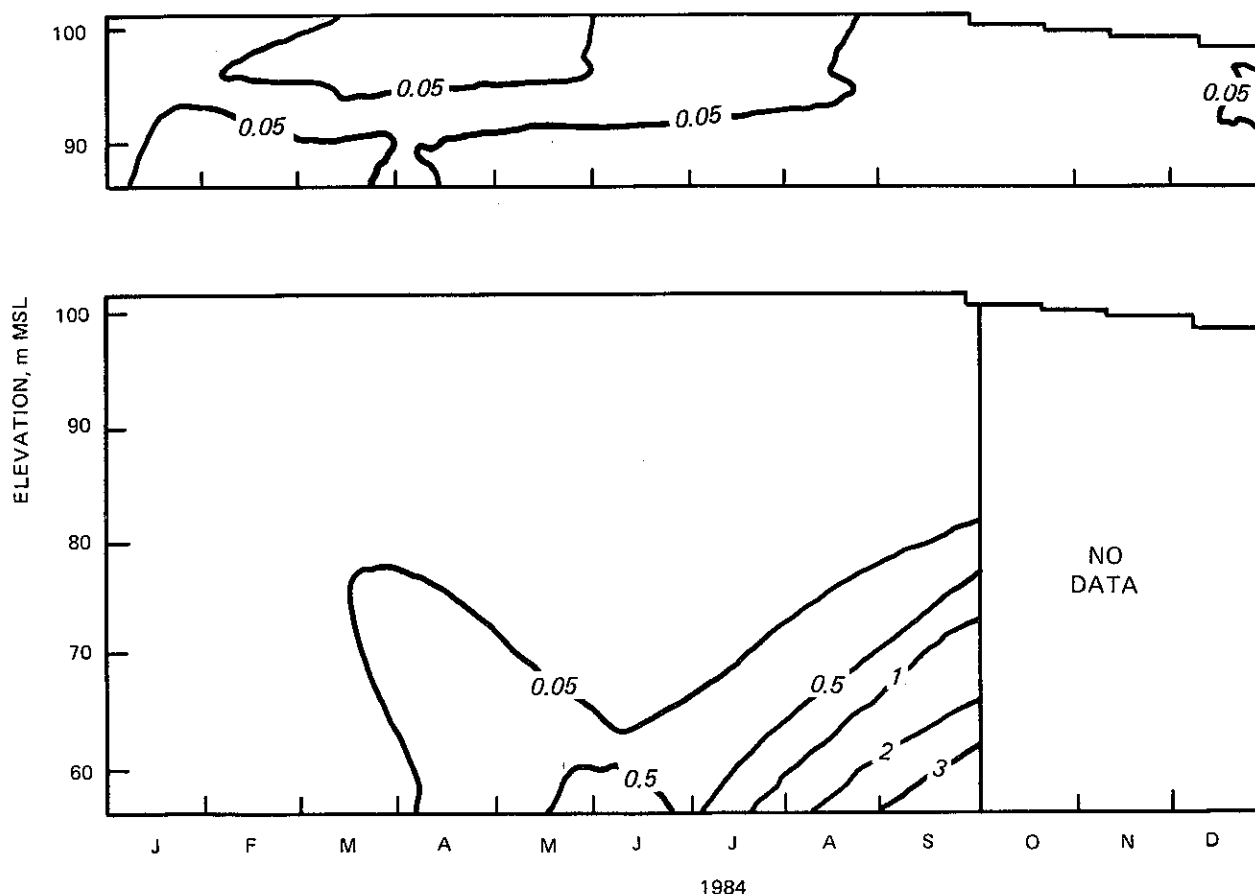

Figure 58. Temporal and vertical patterns in total manganese (mg Mn/l) at Station 40 (upper panel) and at Station 20 (lower panel) during 1984

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Total sodium, total potassium, total calcium, and total magnesium were determined on 5 March, 3 April, and 11 September. In general, concentrations were low, and few seasonal and spatial fluctuations were apparent. Total sodium ranged from 2.3 to 3.6 mg Na/l, potassium from 1.2 to 2.1 mg K/l, calcium from 1.3 to 2.6 mg Ca/l, and magnesium from 0.9 to 1.4 mg Mg/l.

Total alkalinity and hardness were characteristic of systems having a low buffering capacity. Total alkalinity from March through June ranged from 6.8 to 8.2 mg $CaCO_3$/l, and minimal spatial and vertical differences were reported. However, in September, total alkalinity increased to 10-13 mg $CaCO_3$/l at a majority of the depths at each station. Highest concentrations were observed at Stations 20 and 30, where hypolimnetic values of 22 and 17 mg $CaCO_3$/l, respectively, were reported. Hardness displayed a seasonal range of 7.2 to 12.3, and spatial and vertical differences were not pronounced.

Sulfate concentrations were low throughout Clarks Hill Lake. Sulfate ranged from less than 5.0 mg SO_4/l to undetectable levels, and longitudinal patterns were minimal. Late in the stratified period, sulfate concentrations declined to undetectable levels in the hypolimnion at Stations 20 and 30. This observation may be related to the occurrence of low dissolved oxygen concentrations and bacterial reduction of sulfate to sulfide. However, sulfide could not be detected in the bottom waters of these stations with present analytical techniques.

Seasonal and spatial trends in chlorophyll a concentrations for Clarks Hill Lake (Figure 59) were dissimilar to those observed in Richard B. Russell Lake. Although values were generally lowest in winter and highest in summer, exceptions were evident in March and April. Concentrations during this period were highest in the middle and upper reaches of the reservoir, and longitudinal gradients were evident toward the dam. For instance, chlorophyll a concentration was 14.7 mg/cu m at Station 30 but only 2.16 mg/cu m at Station 20. Richard B. Russell Lake, on the other hand, exhibited low concentrations throughout the lake during the winter period. Concentration increased during the summer months and was generally highest at Stations 30 and 40 during this period. Peaks in concentration of 12.1 mg/cu m in July and 9.7 mg/cu m in September were observed at Station 40. High inflow velocities appeared to influence concentration in the tailwater area (i.e., Station 50), as values were low throughout the year.
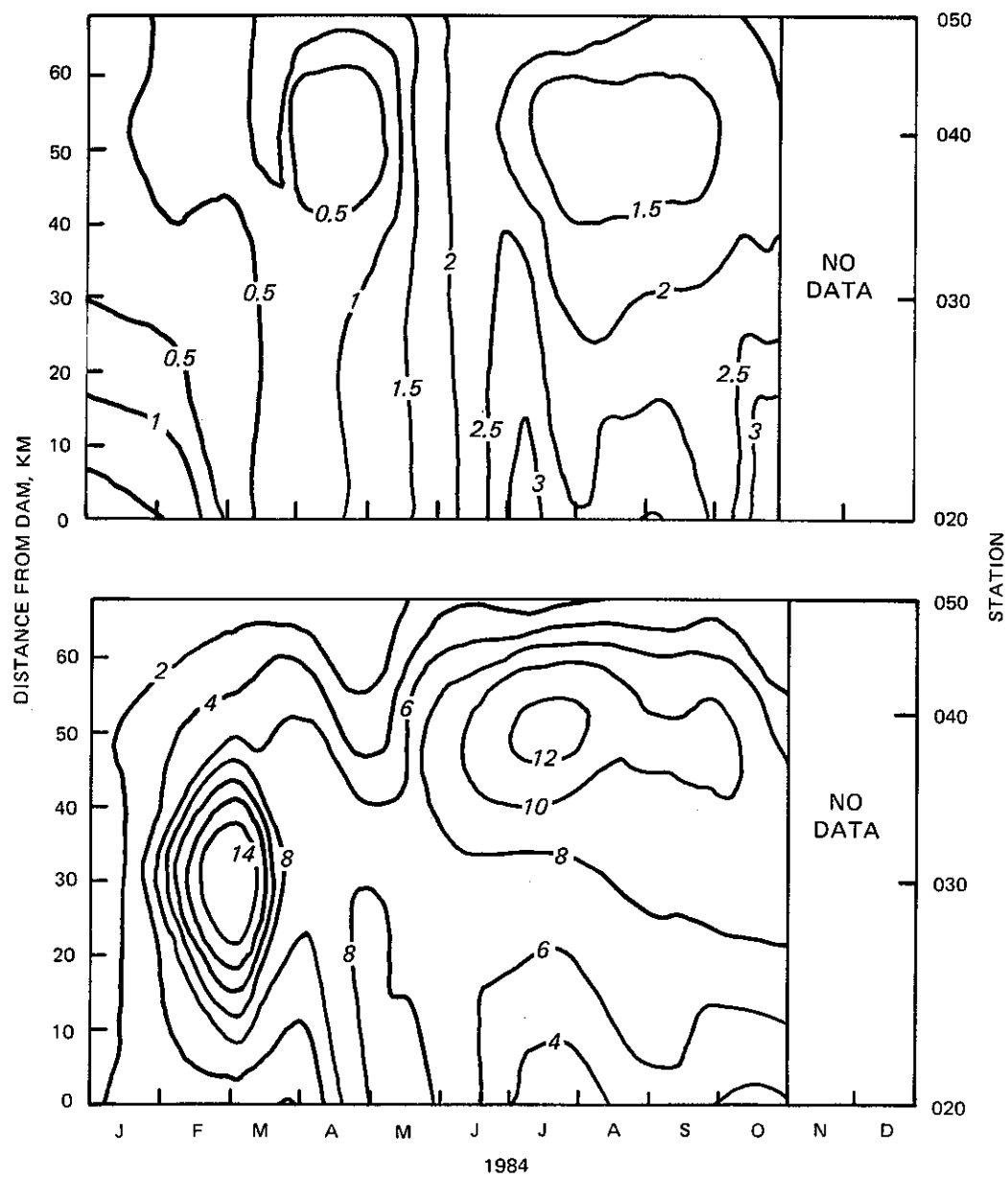
Figure 59. Temporal and longitudinal patterns in Secchi disc (meters;
upper panel) and chlorophyll a concentration (mg/cu m; lower panel)
for the main basin of Clarks Hill Lake

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Interactions Between Releases and Receiving Waters

Introduction. Interactions between reservoir releases and receiving
waters can be assessed using several techniques. One method, which involves
the frequent measurement of temperature and oxygen profiles over several days,
can be used to infer the influence and placement of reservoir release water
in the downstream reservoir. However, since temperature and oxygen profiles
can be influenced by factors within the downstream reservoir as well as by
the upstream discharge cycle, an independent estimate of the influences of
reservoir releases can be made by dye-injection studies. Time of travel,
water movements, entrainment effects, and plunging flows can be more directly
measured by following the progress of dye injected into the inflowing water
(Johnson, 1984). Direct velocity measurements made with a velocity meter
can also be used to corroborate estimates of flow made from dye studies.
All three methods were used to assess release-water/receiving-water interac-
tions in both Clarks Hill and Richard B. Russell Lakes. Information gained
through the conduct of these studies provides insight to physical factors
influencing water quality dynamics in these two lakes.

Diel studies. Diel studies were conducted in the headwater areas of
Clarks Hill and Richard B. Russell Lakes during the periods 13-15 August and
13-14 August 1984, respectively. These studies involved the repeated in-situ
sampling of several selected stations in the extreme upstream reach of each
lake. Vertical profiles of dissolved oxygen, temperature, pH, and specific
conductance were obtained at stations and time intervals indicated in
Tables 9 and 10. Five sampling rounds were conducted at eight stations in
Clarks Hill Lake, starting at 0725 hr on 13 August and ending at 1120 hr on
14 August. One additional sampling round was conducted in Clarks Hill Lake
from 1145-1300 hr on 15 August to better define longer term temporal changes.
Nine sampling rounds were conducted at 10 stations in Richard B. Russell Lake,
starting at 0748 hr on 13 August and ending at 1642 hr on 14 August.

Releases from Hartwell and Richard B. Russell Dams had been suspended
for approximately 36 hr (in accordance with normal operating schedules) over
the weekend of 11-12 August. Releases from the tainter gates at Richard B.
Russell Dam were initiated at about 0830 hr on 13 August, and a continuous
flow of 7,000-9,500 cfs was maintained throughout the study period. Releases
from Hartwell Dam were in accordance with normal operational procedures and

were dictated by peak demands for power generation. Unlike the near-surface releases experienced at Richard B. Russell Dam, releases from Hartwell Dam were withdrawn from the lower portion of the water column. As a result, Hartwell releases were lower in temperature and dissolved oxygen than releases from Richard B. Russell Dam.

Temperature measurements at selected stations in the headwater region of Richard B. Russell Lake are shown in relation to changes in Hartwell Dam discharges in Figure 60. Contour plots of temperature (Figures 61 to 63) and oxygen (Figures 64 to 66) distribution in Richard B. Russell Lake are shown for nine sampling rounds. The initial temperature profile in the headwater region of Richard B. Russell Lake (Round 1; Figure 61) indicated stratified thermal conditions extending upstream past Station 180, with warmer conditions in the surface toward the main body of the reservoir. The gradual upward tilt in the temperature isoclines toward Hartwell Dam probably reflects the result of enhanced stratification downstream toward the main reservoir body, as well as a return of downstream water to the headwater region following a weekend without generation flows. Round 2 (Figure 61), conducted at about 1200 hr, documented the results of cooler water from Hartwell releases entering the upper end of Richard B. Russell Lake; data for downstream areas reflect the start of the diurnal heating cycle. By about 1600 hr (Round 3; Figure 61), Hartwell release water extended midway between Stations 180 and 190 and the peak of the diurnal heating cycle was evident toward the main body of the reservoir with a surface temperature near 30 deg. C toward Station 160. At approximately 2200 hr (Round 4; Figure 62), the beginnings of interflow were suggested by profile data near Station 180. Night cooling was evident downstream of Station 180. The generation cycle for 13 August ceased at about 2200 hr. Round 5 (Figure 62) temperature data show the 18-deg. isocline compressed toward the surface suggesting further movement of cooler water downstream beneath warmer surface water. Round 6 (Figure 62) shows the 16-deg. isocline slightly upstream of its location in Round 5, suggesting movement of water back upstream after cessation of flow. (This trend is also evident in Figure 60, which shows increasing temperatures at Stations 190 and 182 during the morning hours of 14 August.) Composite results for the second day of sampling (Rounds 7-9; Figure 63) are less clear. Between Rounds 6 and 7, the 16-deg. isocline moved back downstream, although generation did not start until 1100 hr. It is possible that seichelike water movement as a result of

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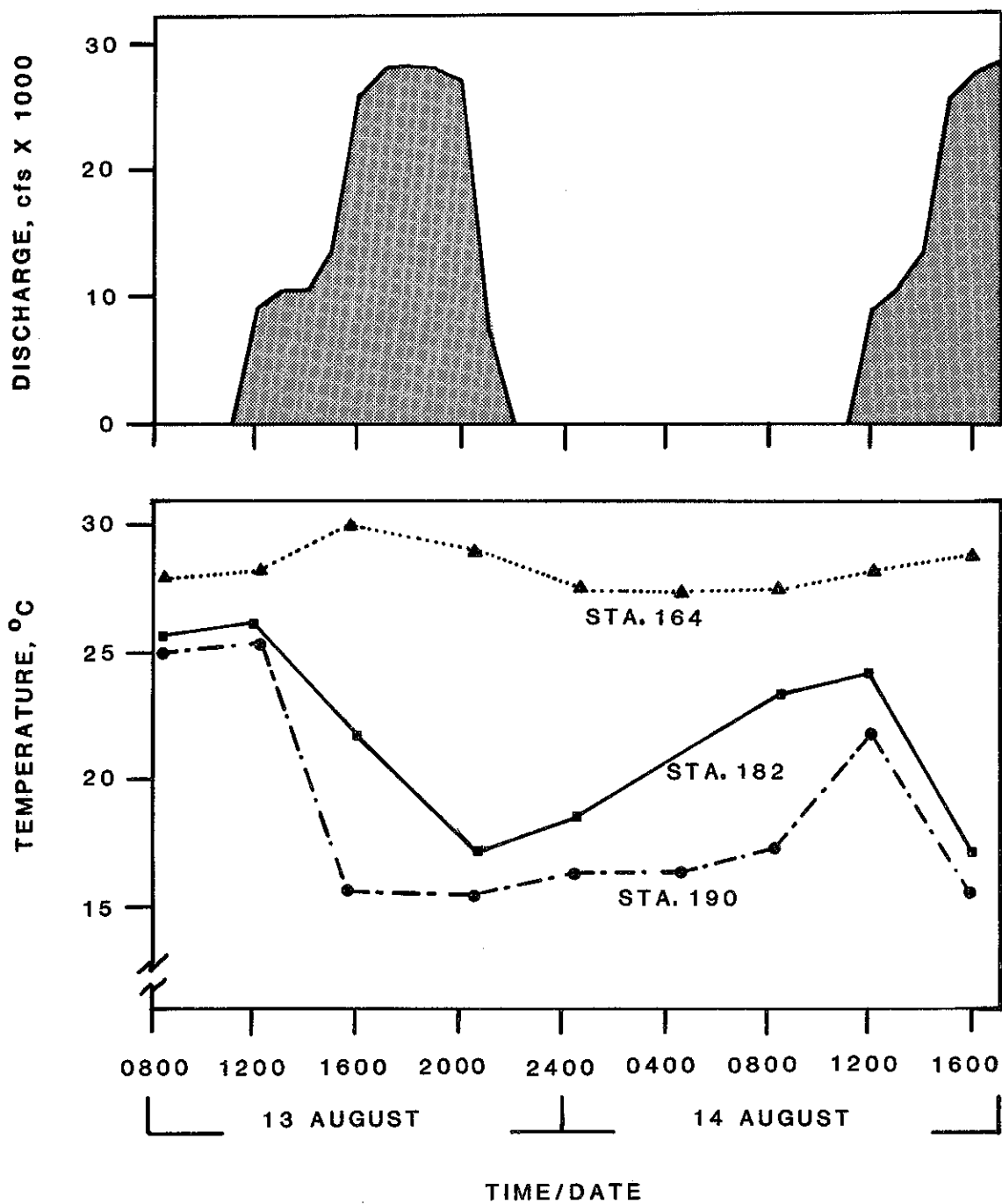
Figure 60. Changes in Hartwell Dam discharges (upper panel)
and surface temperatures (deg. C) at Stations 164, 182, and
190 in Richard B. Russell Lake
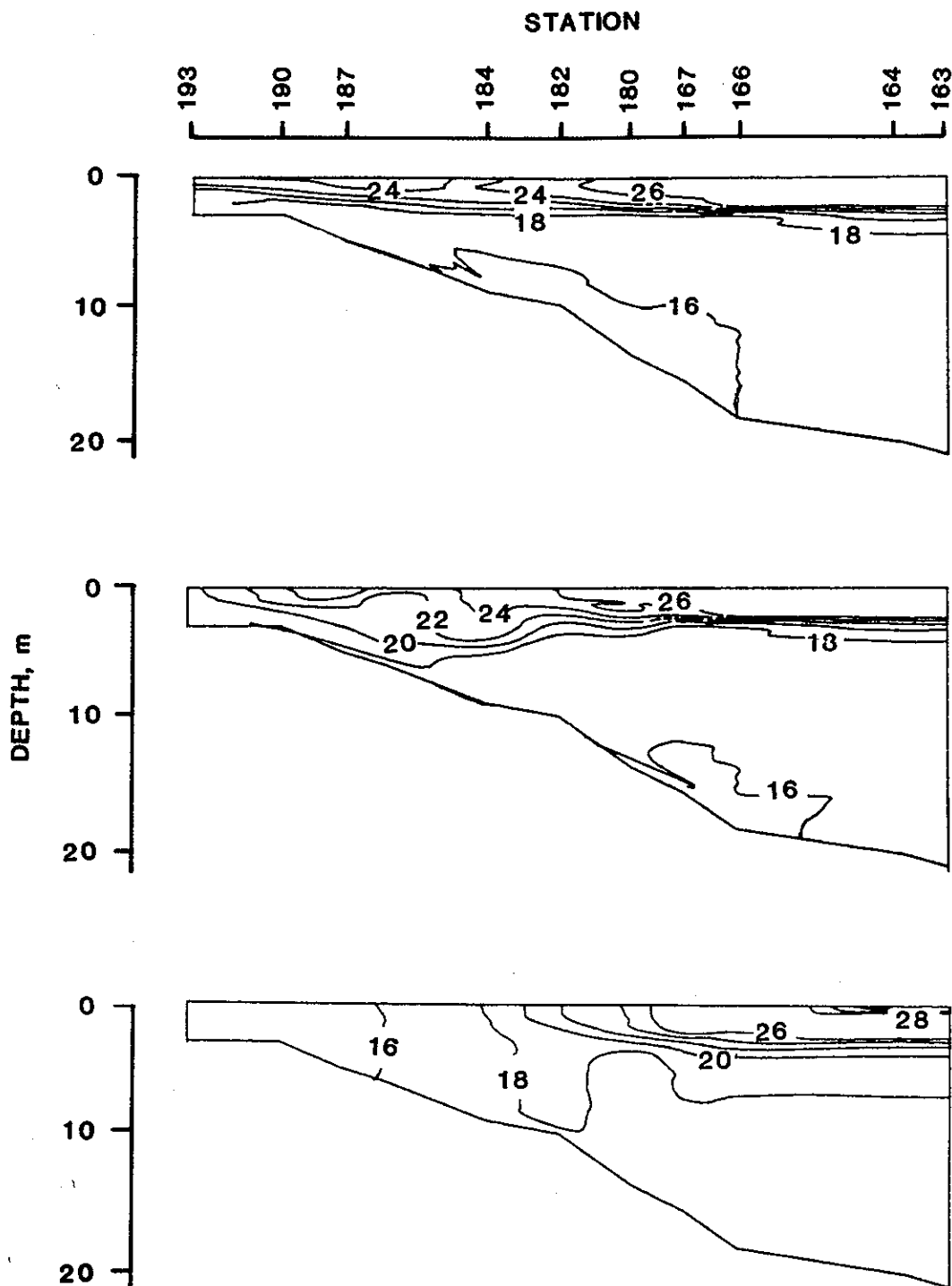
Figure 61. Longitudinal and vertical distribution in temperature
(deg. C) in the headwater region of Russell Lake for Round 1 (up-
per), Round 2 (middle), and Round 3 (lower panel)
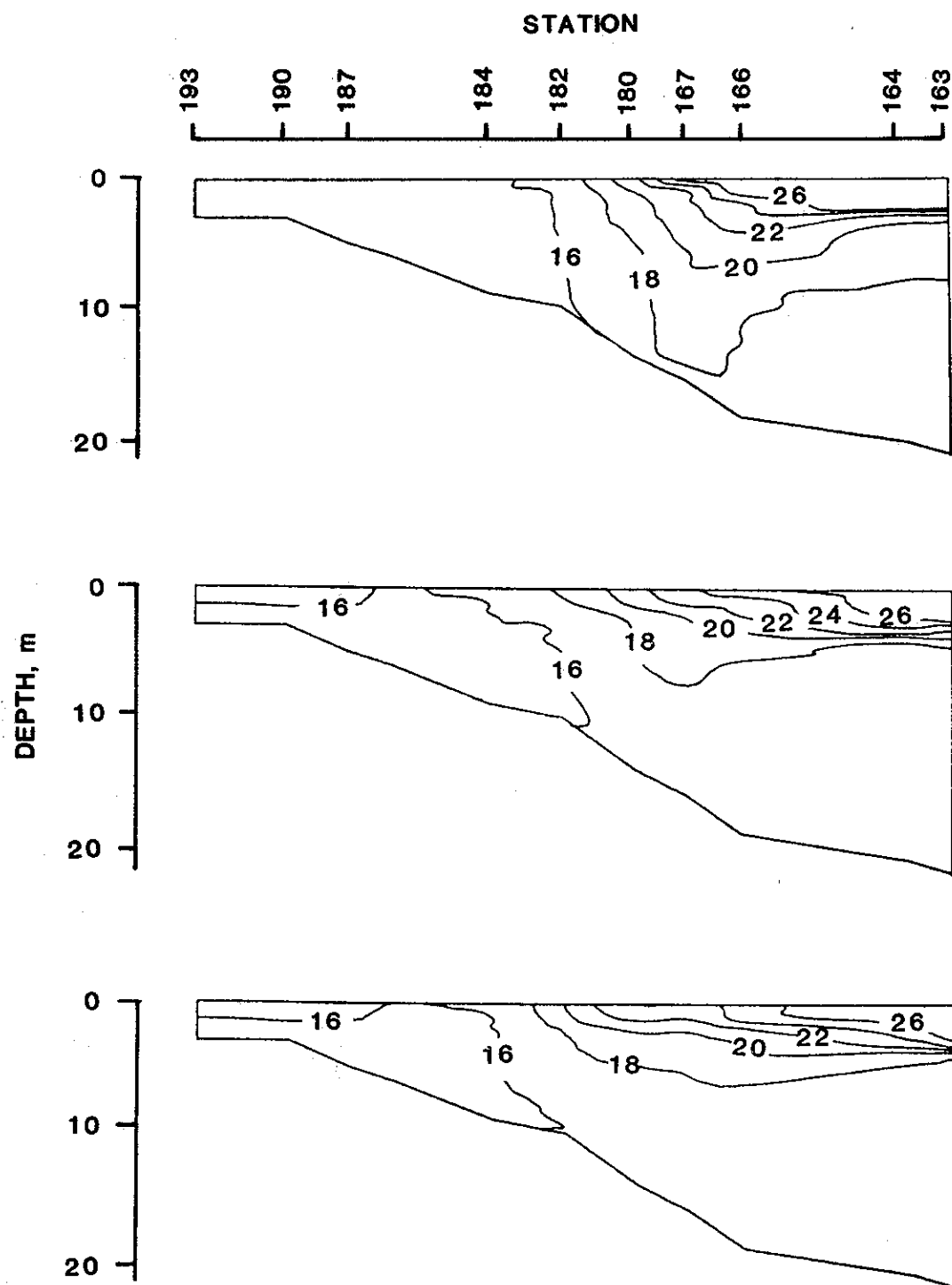
Figure 62. Longitudinal and vertical distribution in temperature (deg. C) in the headwater region of Russell Lake for Round 4 (upper), Round 5 (middle), and Round 6 (lower panel)
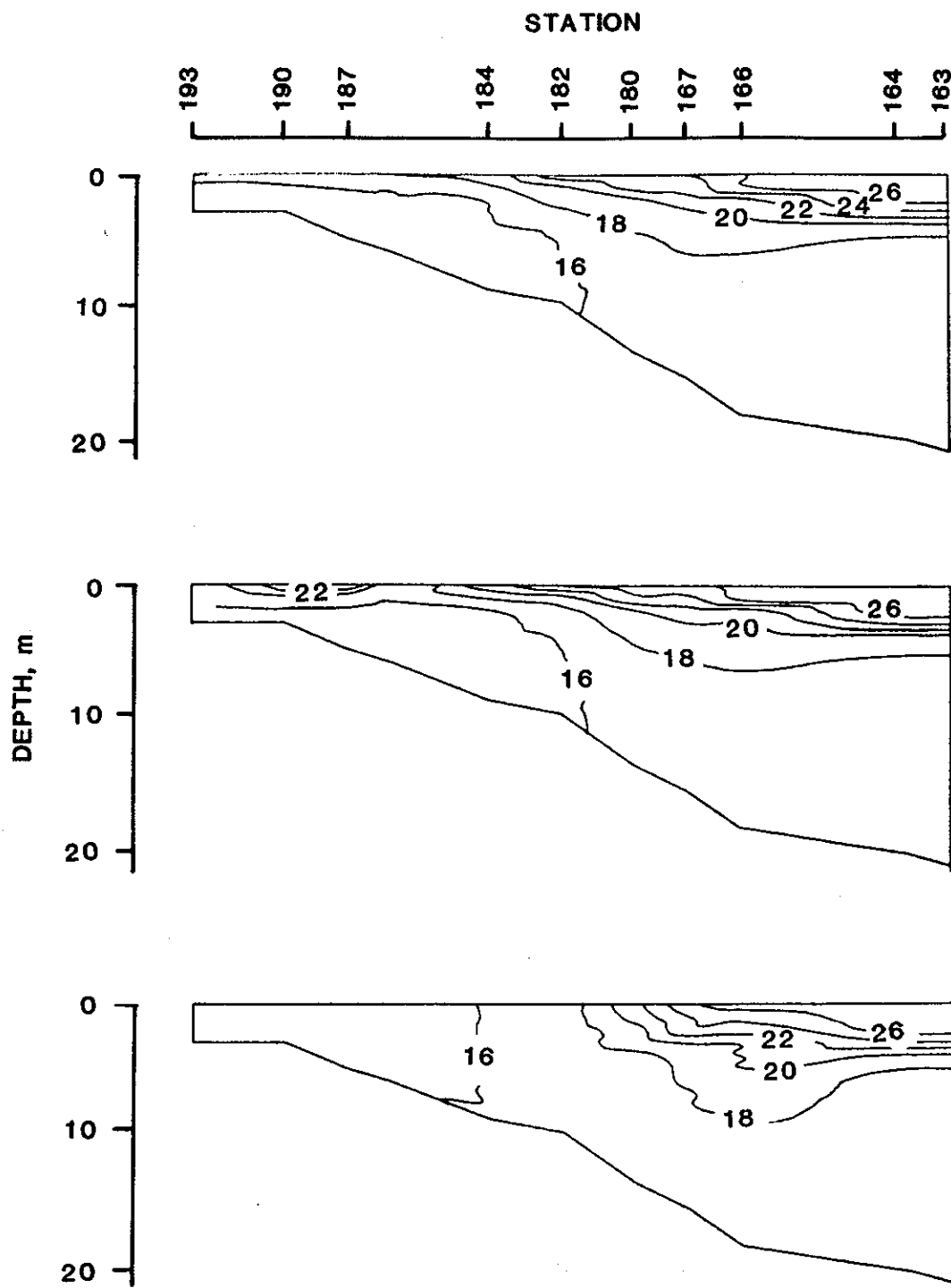
Figure 63. Longitudinal and vertical distribution in temperature (deg. C) in the headwater region of Russell Lake for Round 7 (upper), Round 8 (middle), and Round 9 (lower panel)
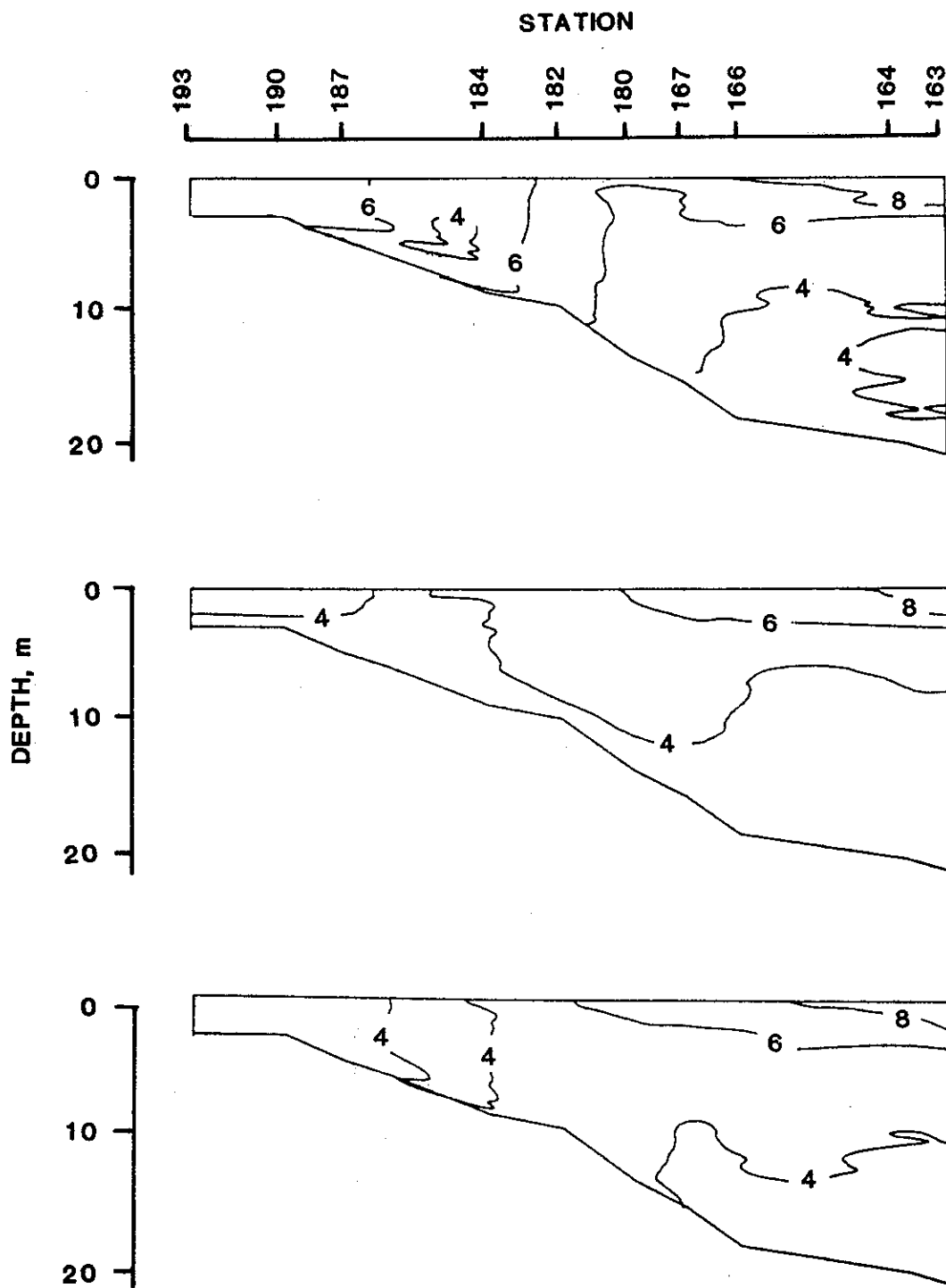
Figure 64. Longitudinal and vertical distribution
in dissolved oxygen (mg/l) in the headwater region
of Russell Lake for Round 1 (upper), Round 2 (middle), and Round 3 (lower panel)
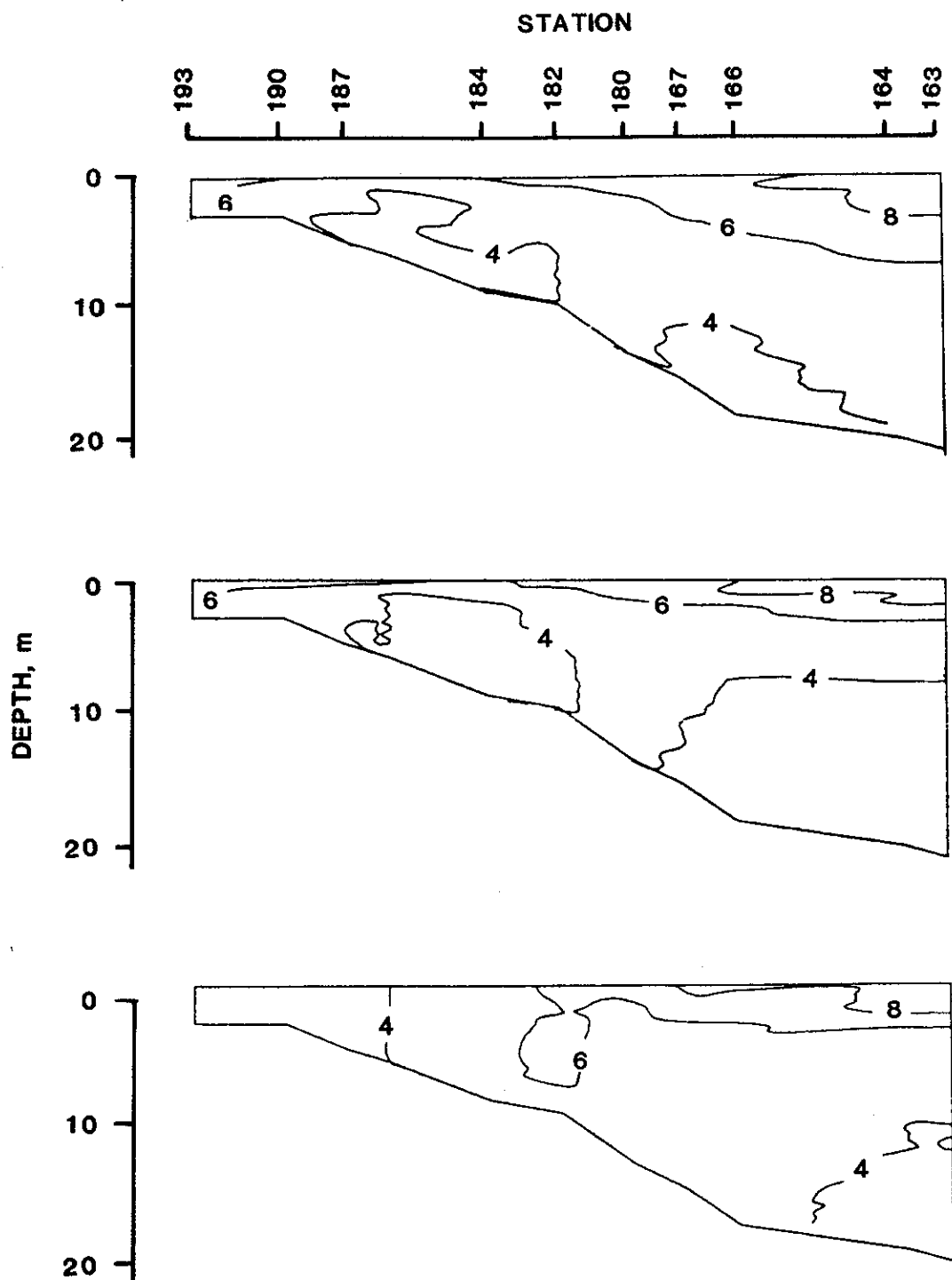
Figure 65. Longitudinal and vertical distribution in dissolved oxygen (mg/l) in the headwater region of Russell Lake for Round 4 (upper), Round 5 (middle), and Round 6 (lower panel)
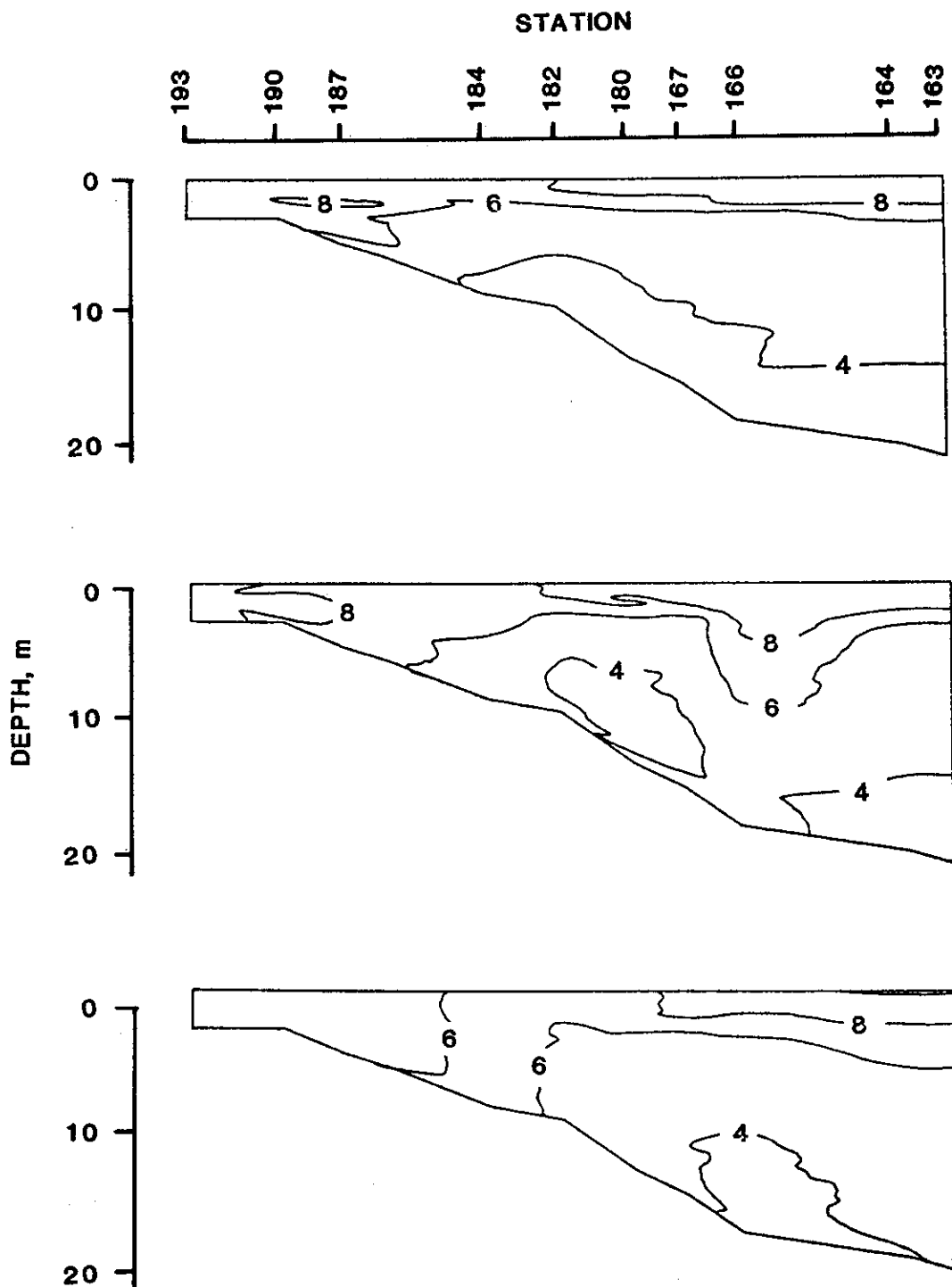
Figure 66. Longitudinal and vertical distribution
in dissolved oxygen (mg/l) in the headwater region
of Russell Lake for Round 7 (upper), Round 8 (mid-
dle), and Round 9 (lower panel)

sudden flow cessation could have resulted, although there is no corroborating evidence of this phenomenon. There was essentially no change between Rounds 7 and 8.

Dissolved oxygen profiles (Figures 64 to 66) indicated similar phenomena occurring as indicated by changes in temperature profiling. Prior to generation (Round 1; Figure 64), oxygen values near saturation occurred near the surface in the entire headwater region. Values near 4 mg/l occurred below a depth of about 10 m as a result of normal summer depletion under stratified conditions. Profiles during Round 2 (Figure 64) showed little evidence of release water intrusion, but there was some evidence of diel biological production toward the downstream stations as indicated by the slightly elevated oxygen readings in the surface waters. Profiles collected during Round 3 (Figure 64) indicated a decrease in oxygen content in the headwater region as Hartwell release water moved downstream. This was especially evident by Round 4 (Figure 65) when water with an oxygen concentration near 4 mg/l had reached past Station 190. This trend continued slightly with Round 5 (Figure 65). As with temperature data, oxygen results with Rounds 6-9 (Figures 65 and 66) are unclear. There appears to have been some movement both upstream and downstream. By Round 9, however, water low in oxygen from the second day of generation had reached past Station 190.

Results of the diel study in Richard B. Russell Lake headwaters indicate movement of release water as a plug flow past Station 190 to a distance of at least 10 km from Hartwell Dam. At this location there appears to have been the formation of an interflow, although the cyclical nature of the generation cycle makes inferences from temperature data difficult because of complex water movements. The data indicate some upstream water movement during the nongeneration period, followed by continued downstream movement during the next generation cyle.

Diel sampling in Clarks Hill Lake documented less dynamic changes in temperature and oxygen conditions (Figures 67 and 68). Since discharges from Richard B. Russell Lake were continuous surface withdrawals rather than cyclical, peaking power generation schedules, there was a gradual and continuous movement of slightly cooler but more oxygenated water down into Clarks Hill Lake. Data for Round 1 indicated uniform conditions from Station 40 to Station 50. The influence of Richard B. Russell Dam releases was evident from Round 2 data, which indicated somewhat cooler (20 deg. C) water pushing
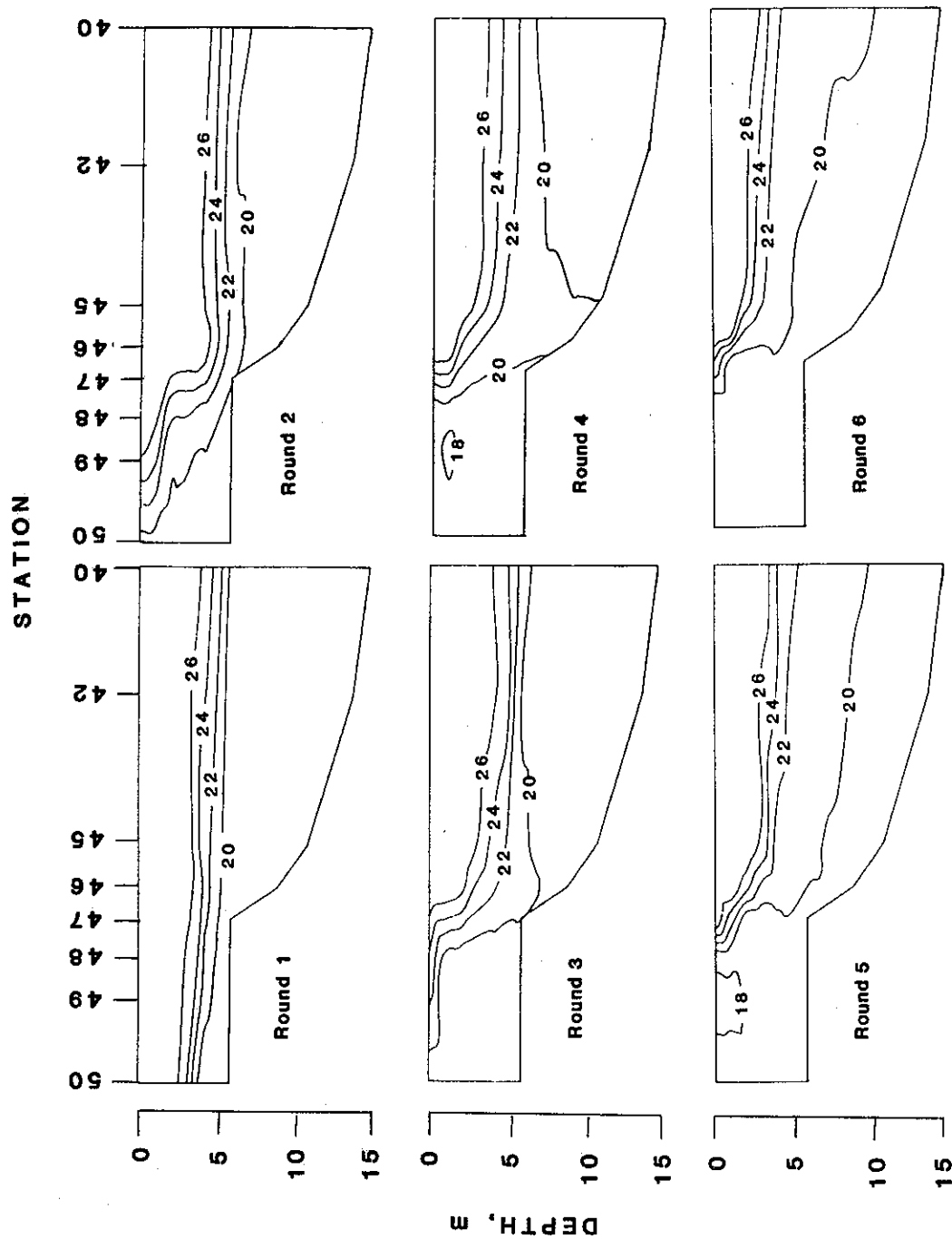
Figure 67. Longitudinal and vertical distribution in temperature (deg. C) in the headwater region of Clarks Hill Lake for Rounds 1-6
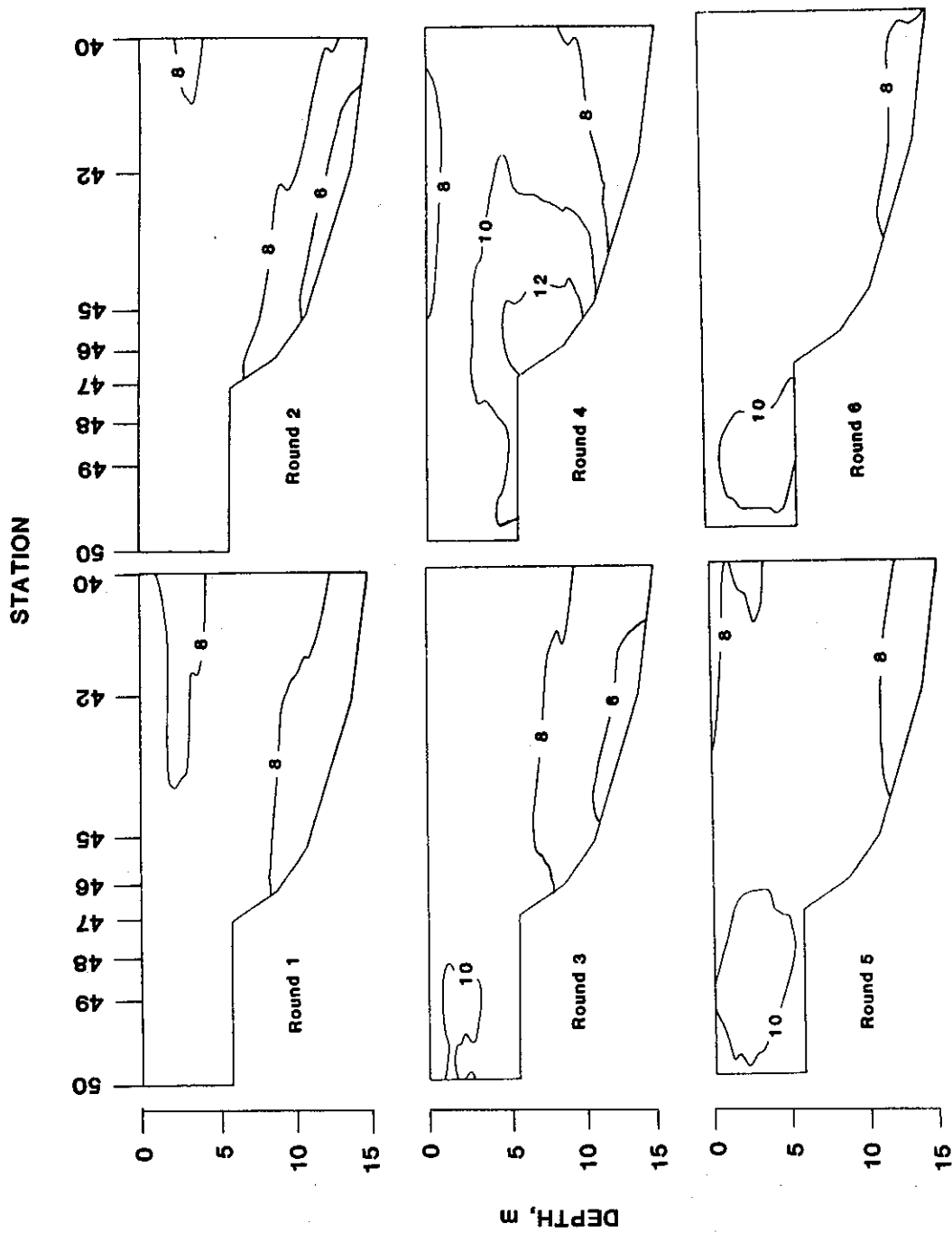
Figure 68. Longitudinal and vertical distribution in dissolved oxygen (mg/1) in the headwater region of Clarks Hill Lake for Rounds 1-6

downstream into the headwater region of Clarks Hill Lake. Since release water was aerated upon release, slightly supersatured water was found in the headwater region. This slightly cooler and more oxygenated water gradually moved further downstream, as shown by Rounds 3 and 4. By the following day (Round 5), the cooler discharge water appeared to be moving as an interflow beneath the warmer surface water. Further evidence for this was inferred from Round 6 temperature data from the third day, which indicated that additional cooler water had moved toward the surface (note the upward progression of the 20-deg. isocline). Oxygen profiles indicated a decline in the amount of supersaturation since slightly lower values occurred at downstream stations. Oxygen values remained slightly elevated near Station 50 as a result of continuous input of aerated Russell Dam release water.

Dye-injection study. Rhodamine WT, a commonly employed water-tracing agent (Johnson, 1984), was injected in the Hartwell Dam tailwater area in an effort to trace water movements and calculate times of travel for parcels of water released to Richard B. Russell Lake during a generation cycle. Subsequent sampling of the moving dye mass allowed documentation of vertical and horizontal movement of water first sampled at the time of dye injection. Assumed in such assessments is the transport of the dye coincident with the movement of the original water mass.

An open, plastic-lined barrel containing approximately 25 gal (94.75 l) of Rhodamine WT dye was positioned near midchannel, approximately 0.8 km downstream from the Hartwell Dam, just prior to the initiation of generation at 1100 hr on 20 August. The barrel was manually toppled from shore, using a line attached to the lid, and the dye escaped and mixed with release water. The release of dye occurred at approximately 1115 hr. During the following 48-hr period, samples for dye concentration were collected at sites downstream in an effort to locate the dye. Sampling involved two boats each equipped with Turner Design Fluorometers. Samples were pumped from selected depths, passed through the flow-through cell of the fluorometers, and values of relative fluorescence were related to those of standard solutions to determine dye concentration. Coincident measurements of temperature were obtained using an in-line thermistor. Daily measurements of in-situ variables were collected at selected stations from Station 190 to Station 160 to provide reference information concerning general physical and chemical conditions in the study area. These measurements were obtained by a third sampling boat working independently.

Initial attempts at defining the distribution of dye were complicated by the rapid movement of water in the upstream reaches of the study area (i.e., above Station 190). Because of this problem, sampling consisted of fixed-station monitoring of dye concentration in an effort to calculate time-of-travel of the moving water mass during the first day of the study. The times of observance of maximum dye concentration at selected stations were recorded on four separate occasions during the first 9-hr period following release of the dye. This time period encompassed the entire generation cycle. Presented in Figure 69 are discharge rates and distances downstream of maximum dye concentrations at various times following the initiation of discharge.
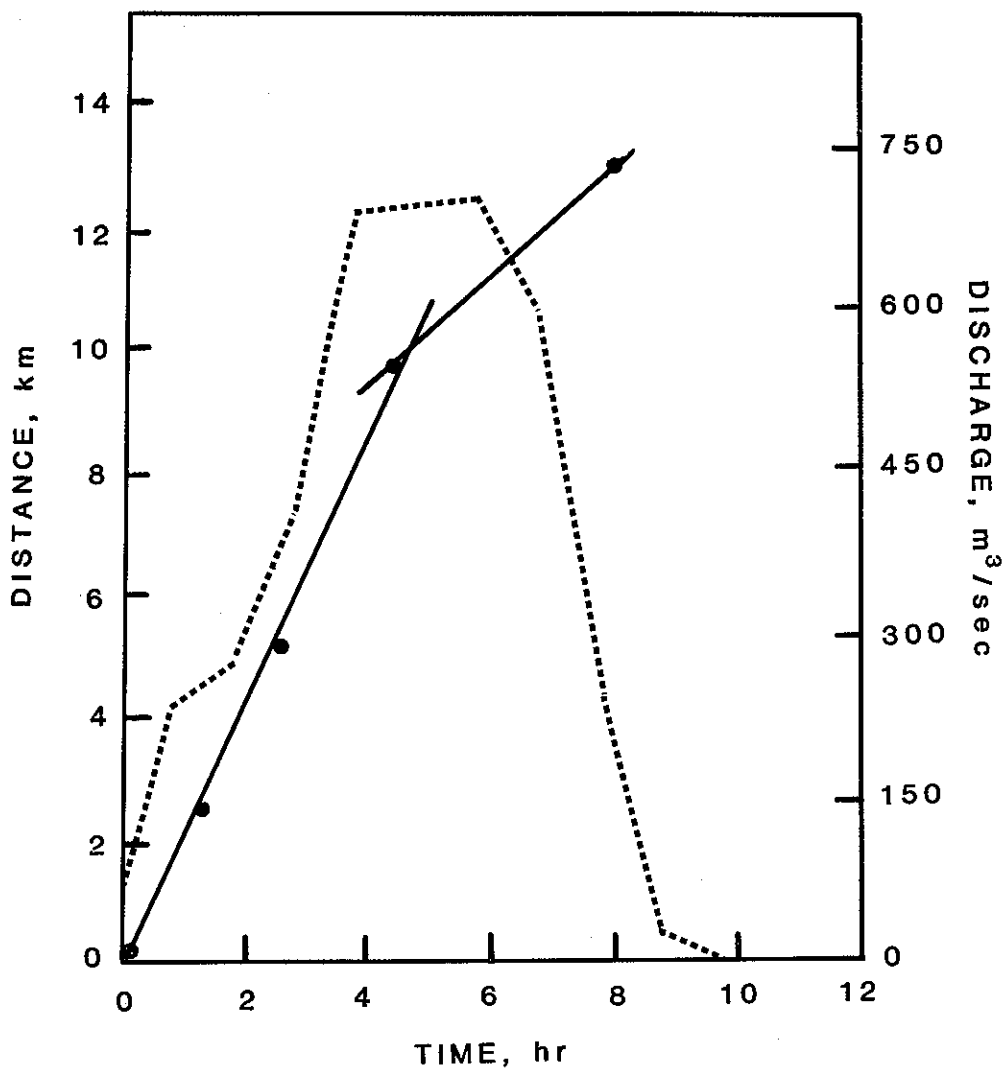
Figure 69. Discharge from Hartwell Dam (cms) and distance of dye movement at various times during the first 9-hr period of release on 20 August 1984

The relationship between distance and time describes the average velocity of the dye (and therefore, the water). Velocity during the first 4-hr period, which corresponded with the rising limb of the release hydrograph, averaged 2.08 km/hr. This value corresponds well with the average value obtained through in-situ measurement of velocity using a flow meter (see below). Average flow during the following 4-hr period (i.e., during the fall in the release hydrograph) averaged approximately 0.81 km/hr.

The maximum concentration of dye was located immediately upstream of Station 180 and approximately 3 km downstream of Station 180 on 21 and 22 August, respectively (Figure 70). The vertical distribution of dye on the morning of 21 August documents the occurrence of a pronounced interflow of release waters at a depth of 4-5 m and the presence of a plunge point approximately 13 km downstream from Hartwell Dam (Figure 71). Such an occurrence was related to differences in the densities of inflowing and lake waters. The placement of inflows below the surface of Richard B. Russell Lake is of potential importance since material loads, particularly nutrients, would be partially unavailable for phytoplankton in the upper, well-lighted strata of the lake. The containment of inflows to a relatively narrow strata would also promote more rapid movement of these waters downstream within Richard B. Russell Lake.

Velocity measurements. Direct measurements of velocity profiles were obtained in Richard B. Russell Lake headwaters in conjunction with the dye-injection study as an independent assessment of water movements during generation releases from Hartwell Dam. Velocity profiles were measured with a Marsh-McBirney Model 201M electromagnetic current meter. The probe was suspended in the water column and maintained in the direction of flow with the use of a 33-kg finned weight. Readings were taken every meter down to a maximum of 10 m. Measurements of velocity were not reliable below about 10 cm/sec (0.4 km/hour) as it was not possible to ascertain that the probe was being maintained parallel to the direction of flow. Three sampling rounds were made as tabulated in Table 10, but as nearly all measurements for Round 3 were below the reliable detection limit, those results are not included in this analysis. Results for the first two rounds are shown in Figure 72.

Round 1 indicates the longitudinal velocity profiles approximately 2-3 hr following start of the generation cycle on 20 August. Flow velocity was 2.5 km/hr near the surface of Station 190 and ranged from 0.4 to 0.9 km/hr

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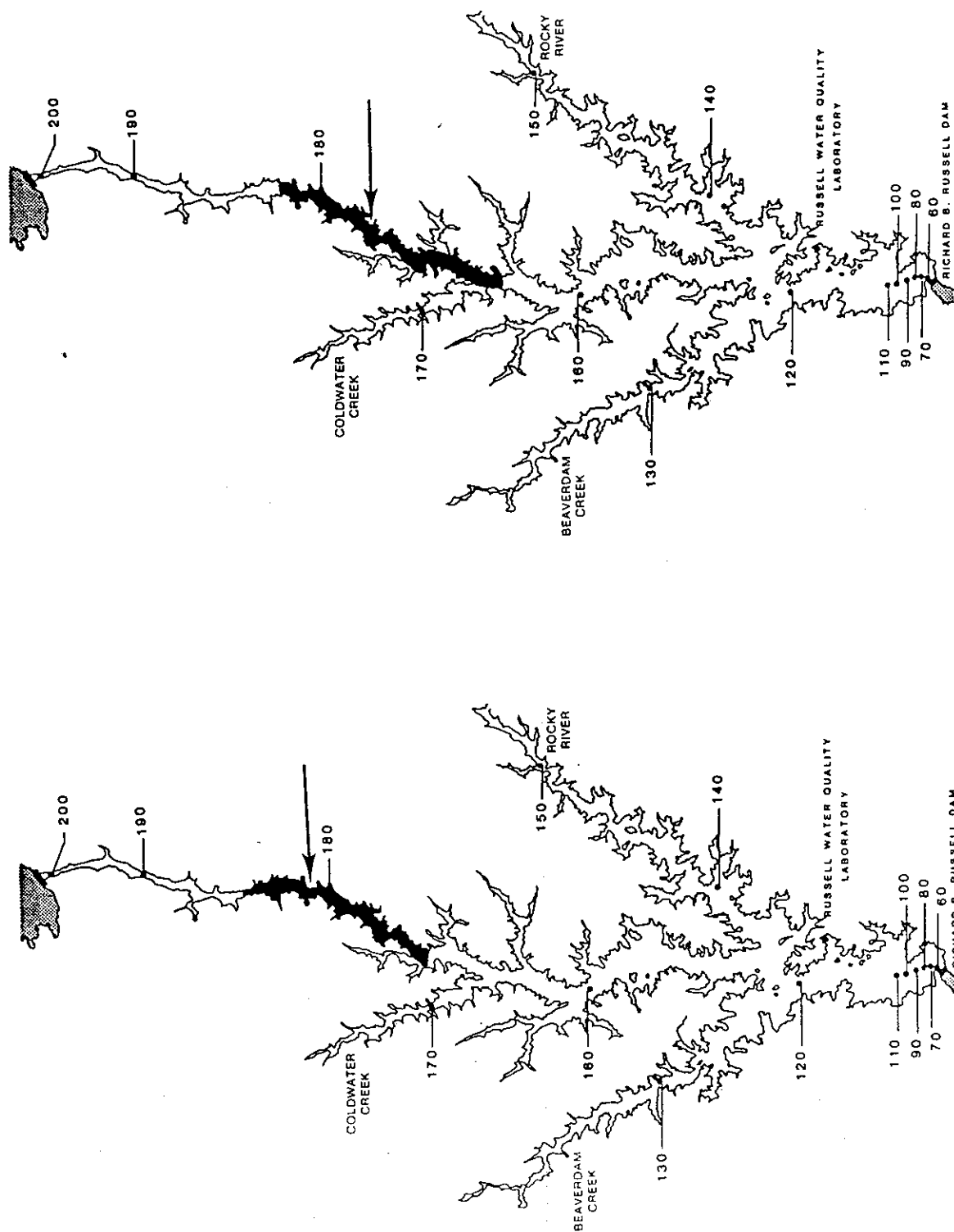
Figure 70. Location of maximum dye concentration
(shaded area) on 21 and 22 August 1984
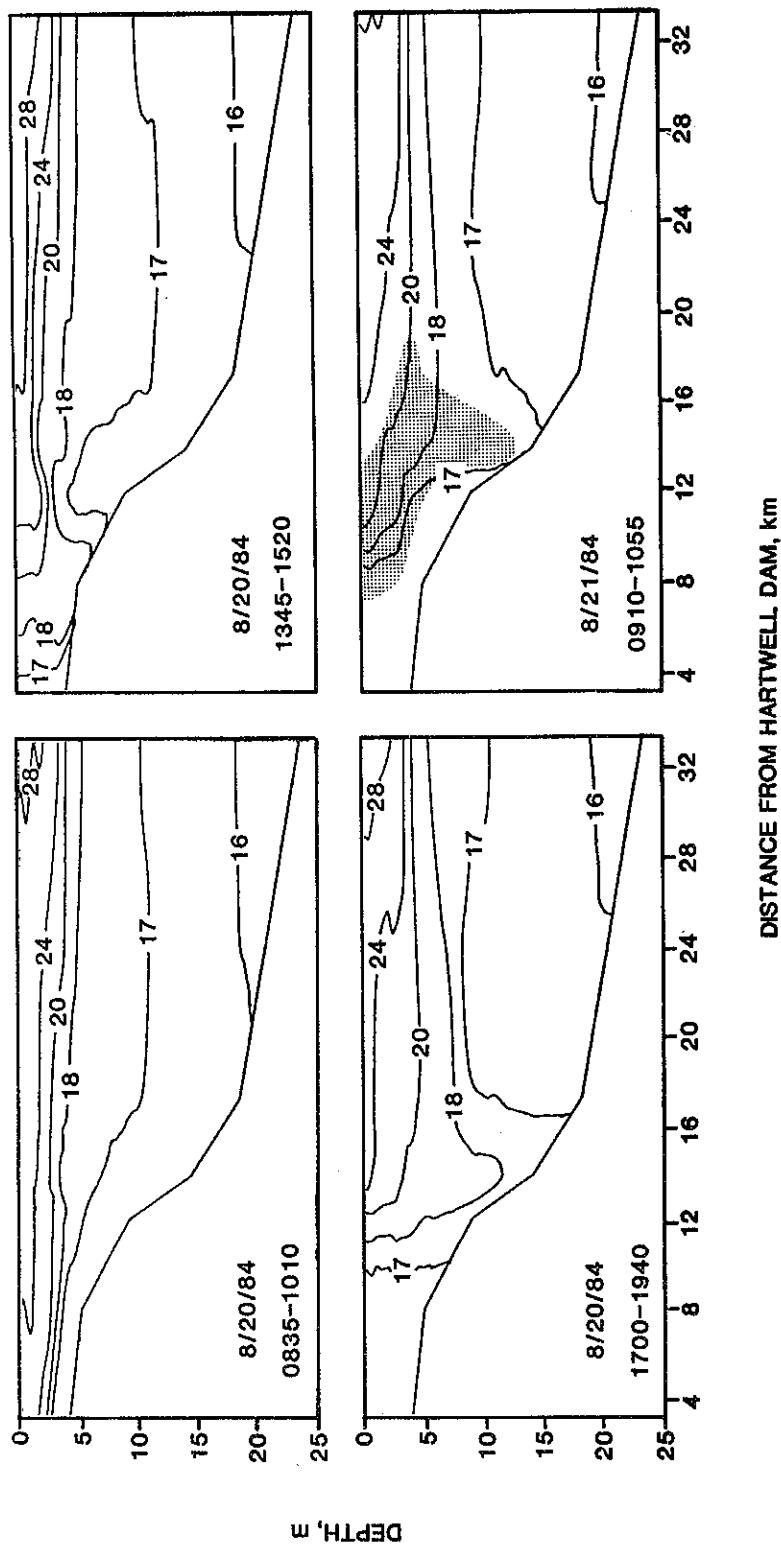
Figure 71. Vertical and longitudinal distribution of temperature in the headwater region of Russell Lake on 21 and 22 August 1984. Shaded area represents area of maximal dye concentration

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DISTANCE FROM HARTWELL DAM, km
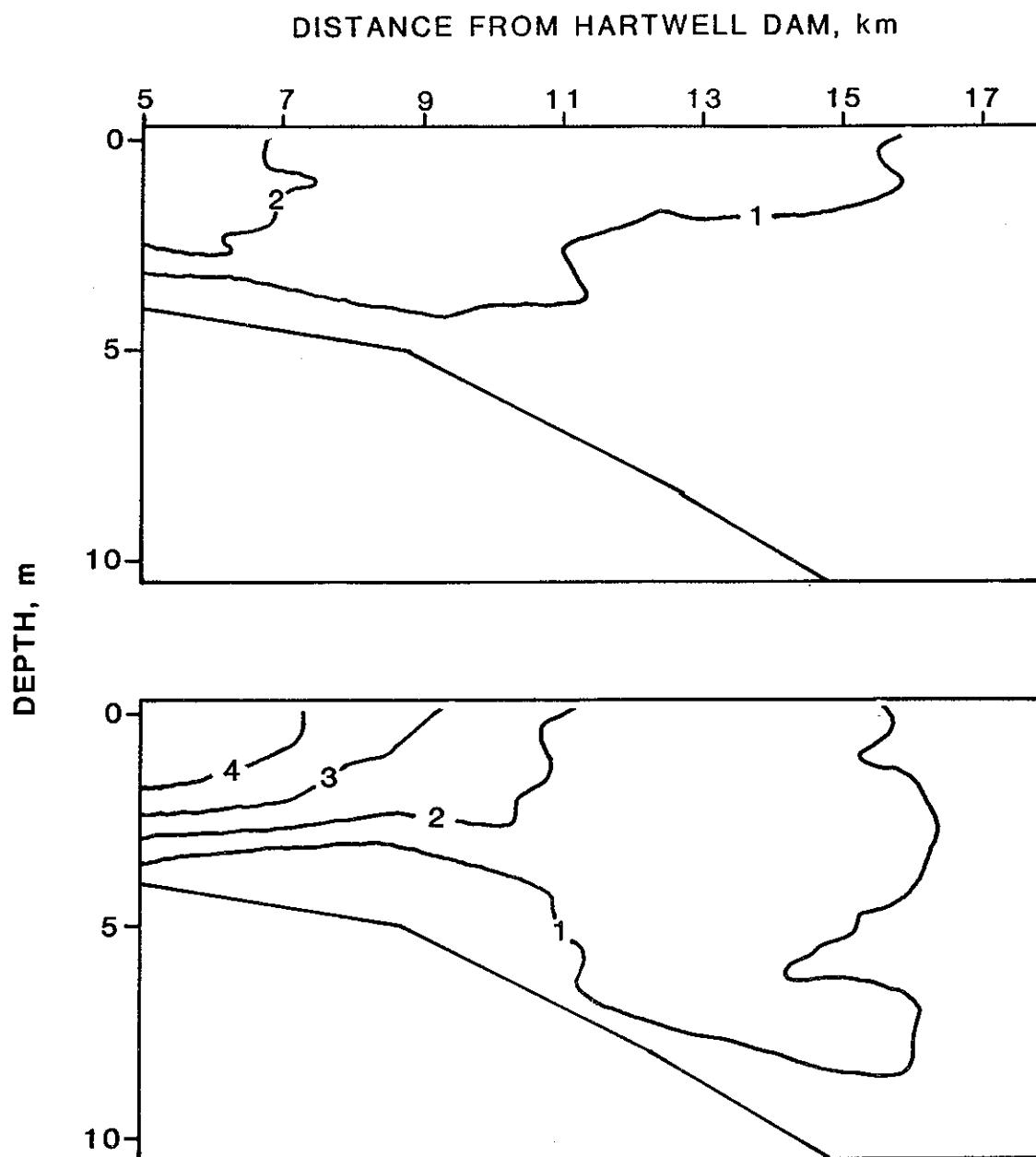
Figure 72. Vertical and longitudinal variations in
velocity (km/hr) in the headwater region of Russell
Lake for Rounds 1 and 2 on 20 August 1984

in the top 10 m at Station 166. At this time, peak velocities were near the
surface at most stations with the highest values in the top 1 m. Round 2
shows the velocity profiles during the peak of generation flows for 20 August,
which reached 25,000 cfs. Flow velocity was 5 km/hr at the surface of Sta-
tion 190 and ranged from 0.3 to 0.7 km/hr at Station 166. Peak velocities for
Round 2 moved deeper into the water column as the flow moved downstream. Peak

flow occurred near the surface at Stations 190 and 105 but the peak flow for Station 195 occurred at the 4-m depth. Peak flows occurred between 2 and 4 m at Station 180 and between 2 and 3 m at Station 166. These results are consistent with observations of temperature profiles indicating plug flow down to about 10 km below Hartwell Dam with an interflow starting at about that point. Velocity measurements are also consistent with time-of-travel measurements calculated from the dye-injection studies previously discussed.

PART V: GENERAL DISCUSSION AND CONCLUSIONS

Richard B. Russell Lake

Water quality conditions in the Richard B. Russell Lake basin prior
to filling (i.e., Phase I) were dictated by the characteristics of inflow-
ing waters and the existence of riverine conditions along this reach of the
Savannah River. During this period (i.e., Phase I of filling), only a small
pool existed in the forebay area, and much of the river and its tributaries
were confined to existing streambanks. Because of its shallowness and the
relative magnitude of flow from the Savannah River, the residence time of
waters impounded in the vicinity of the breached coffer dike was minimal
and few changes in water quality were observed. Temperatures were nearly
uniform throughout the water column, and dissolved oxygen concentrations
were near saturation. Chemical concentrations were low and similar to con-
centrations measured at upstream locations. This latter observation was
anticipated since little terrestrial vegetation and soils had been inun-
dated before or during this period.

Observed on several occasions during Phase II of filling were vertical
patterns in the distribution of in-situ variables. These patterns reflected
the influences of the coffer dike on flow patterns in the immediate vicinity
of the dam. The dike, although breached at a central location near the thal-
weg, partially impounded inflowing waters. This, coupled with the fact that
discharges during this period were via sluice gates located at depths below
the crest of the coffer dike, led to the preferential withdrawal of surface
waters. Similar impacts of submerged coffer dikes on water quality have
been reported for other reservoirs.

Significant physical, chemical, and biological changes occurred dur-
ing Phase III of filling. This phase was marked by a rapid 27.5-m rise in
pool elevation between December 1983 and mid-February 1984. The result was
a 19-fold and 68-fold increase in pool area and volume, respectively. By
late-February, nearly 7,700 ha of the basin were inundated and pool volume
had reached approximately 70-75 percent of that anticipated to occur at
power pool elevation. This resulted in greater influences from the Beaver-
dam Creek and Rocky River embayments. Chemical concentrations were often
higher at the embayment stations than in the main basin, suggesting that

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material loading was occurring via these secondary tributaries.

Dissolved oxygen concentrations during this period of rapid rise in pool elevation remained near saturation throughout the water column, and isothermal temperatures were recorded. Evidence of dissolved oxygen depletion was noted only during intermittent periods of weak stratification, and then primarily in the two embayments and at their confluence (i.e., Station 120).

Chemical changes during the filling period were minimal compared to those observed later during stratification, despite the inundation of soils and vegetation. However, since significant quantities of vegetation were inundated during this period, potential impacts on the water quality of the pool were great. For instance, leaf litter can break down rapidly during the early stages of decomposition due to leaching of soluble substances. James (1985b) and Shain (1985), using areal photographic methods and conventional mensuration techniques, developed estimates of inundated tree and leaf litter biomass in Russell Lake. Approximately 548 million kilograms of tree biomass (green weight) were left within the basin, and areal estimates ranged from 8.42 to 28.1 kg/sq m (mean areal estimate was 15.4 kg/sq m). Dry weight estimates of inundated leaf litter ranged from 2.74 to 11.0 million kilograms. Therefore, assuming complete leaf litter breakdown, approximately 1.28 to 5.17 million kilograms of organic carbon, 9,040 to 36,300 kg of nitrogen, and 71 to 286 kg of phosphorus could have been released to the system (James, 1985a).

James (1985a) reported a rapid loss of phosphorus from leaf litter incubated in mesh bags on the bottom of Richard B. Russell Lake immediately after filling. Phosphorus loss during the first month ranged from 27 percent of the initial phosphorus content for a hardwood leaf litter mixture to over 80 percent for white oak leaves (*Quercus alba*). Nitrogen losses were also detected for some leaf types during the initial phase of decomposition. Short-needle pine and white oak litter lost approximately 25 percent of their initial nitrogen contents within the first month. Thus, the potential for a rapid release of nutrients to the system existed during the filling period.

In comparison, Crawford and Rosenburg (1980) reported that rapid leaching from experimentally incubated black spruce (*Picea mariana*) needles resulted in a 25-percent loss of dry weight from the needles within a 3-day period. Since approximately 540,000 tons of black spruce needles were inundated during flooding of Southern Indian Lake (Manitoba, Canada) in 1977,

approximately 3,730 tons of organic carbon could have been released to the system in a short period of time.

Gunnison et al. (1984) studied the impact of initial soil and vegetation flooding on water quality characteristics using laboratory chambers and representative preimpoundment soils from Richard B. Russell Lake basin. Their findings indicated that total and dissolved forms of organic carbon, nitrogen, and phosphorus can be rapidly released to the water column under aerobic and anaerobic conditions and warm temperatures (i.e., 20 deg. C). Iron and manganese were released under anaerobic conditions. These data suggested that initial inundation could have a significant effect on water quality conditions, particularly at warmer water temperatures.

The fact that significant increases in the concentrations of organic material, nutrients, and reduced metals did not occur during the filling period was related to temperature, mixing, and flushing rate. Water temperatures during the period were uniform throughout the water column and averaged 8-10 deg. C. Gunnison et al. (1984) reported significant decreases in the rates at which materials were released from basin soils with decreases in the temperature of laboratory reaction chambers. Thus, the cooler temperatures observed in the lake would have led to reduced release rates. Cool temperatures would also have limited bacterial activity on the basin soils, which at warmer temperatures would have played a more important role in decomposition.

Thermal regime and the maintenance of an oxygenated environment also influenced chemical trends observed in Richard B. Russell Lake. Under these conditions, mixing would tend to redistribute and dilute soluble materials released to the water column during initial inundation, resulting in smaller concentration increases per unit volume. In addition, many metals form oxyhydroxide complexes in the presence of dissolved oxygen and remain insoluble until anaerobic conditions are achieved.

The establishment of stratified conditions in Richard B. Russell Lake led to marked lakewide changes in water quality characteristics. Isolation of hypolimnetic waters high in organic material led to extensive dissolved oxygen depletion and eventual anoxia in a major portion of the reservoir. Associated with this was the establishment of a more reducing environment and concomitant increases in soluble nutrients and metals in the bottom waters. By September, a major portion of the hypolimnion was anoxic, and high concentrations of soluble iron, manganese, phosphorus, and sulfide were observed.

Hypolimnetic conditions varied spatially and temporally. Anoxia and associated increases in chemical concentrations developed first in the upstream areas of the two major embayments where hypolimnetic volumes are small, then spread to the deeper areas of the main basin. This spatial and temporal pattern is typical for long, mainstream reservoirs and may be related, in part, to sources of organic material which create an oxygen demand, and to effects of basin morphology (Hannan, 1979; James and Kennedy, 1985; Kennedy and Nix, 1985). The shallow embayment areas, rich in inundated organic material, had higher sediment area:hypolimnetic volume ratios, allowing more water column contact with the sediment. Decomposition of organic material at the sediment/water interface and in the hypolimnetic water column would, therefore, have a more immediate and rapid impact on dissolved oxygen concentrations. Larger hypolimnetic volume and less sediment contact in the forebay area would act to dilute effects of decomposition, resulting in a slower depletion of dissolved oxygen.

It was anticipated that dissolved oxygen depletion would be severe after the establishment of a hypolimnion due to the presence of a large pool of organic material left in the reservoir during filling. Although effects of initial filling on water quality were minimal, thermal stratification resulted in the isolation of bottom waters and depletion of dissolved oxygen through chemical and biological processes.

Studies performed on representative samples of soils and leaf litter obtained from the reservoir documented high chemical and biological dissolved oxygen demands due to the presence of these materials. For instance, soils subjected to laboratory temperatures of 5, 12.5, and 20 deg. C had depletion rates of 0.017, 0.023, and 0.185 mg O_2/l per day and areal oxygen demands of 16.8, 22.7, and 183 mg O_2/sq m per day, respectively (Gunnison et al., 1984). Leaf material incubated at the bottom depth (i.e., 28-m depth) in Russell Lake also exerted high mean areal oxygen demand during the period February to August (James, 1985b). Areal demands for a hardwood leaf litter mixture ranged from 881 to 3,524 mg O_2/sq m/day, which could account for 22 percent to 89 percent of the total hypolimnetic areal demand at Station 130 (see Table 7). Loading of organic material during storm events would have an additional impact on dissolved oxygen dynamics during stratification, particularly in upstream embayment areas located near inflows. Thus, these materials would be expected to substantially influence the hypolimnetic dissolved oxygen demand.

The severity of hypolimnetic dissolved oxygen depletion in Richard B. Russell Lake was also related to the effects of near-surface tainter gate discharges. Since surface withdrawal leads to the establishment of a strong thermocline high in the water column, hypolimnetic volume was great and exchanges between surface and bottom strata were minimal. Furthermore, in-pool flow patterns established at the depth of the withdrawal zone minimized hypolimnetic flushing and potential reintroduction of dissolved oxygen.

Hypolimnetic dissolved oxygen depletion rates were high during the first year. Based on data collected during the first year of impoundment, volumetric depletion rates ranged from 0.11 mg O_2/l per day at Station 60 to 0.51 mg O_2/l per day at Station 130 (Table 5). These rates were higher than those previously estimated (i.e., 0.10 mg O_2/l per day; US Army Corps of Engineers, 1981) and represent an extreme case since the Russell Lake basin was rich in recently inundated labile organic material.

The large magnitude of hypolimnetic dissolved oxygen depletion in Russell Lake was not unexpected during the first year of impoundment, as other newly created reservoirs have exhibited similar trends. For instance, DeGray Lake, impounded in 1969, experienced a large zone of hypolimnetic anoxia in the early years which extended from the dam to the headwater region (30 km). Vegetation and soils were left intact in a major portion of the basin (Kennedy and Nix, 1985). Mouchet (1984) documented extensive hypolimnetic anoxia in tropical reservoir Poza Honda (Ecuador) shortly after impoundment. He attributed this pattern to a lack of site preparation (i.e., removal of vegetation and topsoil).

Hypolimnetic dissolved oxygen patterns in Russell Lake are, however, expected to change during the next several years as inundated organic material becomes depleted and removed from the system. These contentions are supported by long-term observations of hypolimnetic dissolved oxygen patterns in DeGray Lake (Kennedy and Nix, 1985). As mentioned earlier, during the early years of impoundment, anoxic conditions in DeGray Lake were evident in a major portion of the hypolimnetic zone. However, 10 years after impoundment, anoxia was restricted to a small zone in the forebay and a zone in the upper region influenced by inflows. Other hypolimnetic areas remained oxic throughout the stratified period. The areal dissolved oxygen demand was 1,090 mg O_2/sq m/day during the first year following impoundment but decreased to a mean of 599 mg O_2/sq m/day for the years 1972 to 1980 as terrestrial material became exhausted.

Many of the chemical changes that occurred in Russell Lake were related to the establishment of an anoxic hypolimnion which altered the physical conditions of the bottom waters. Inundated bottomlands, vegetation, and sediments clearly represented a large pool of material which could be potentially released to the water column during stratification. These materials appeared to be an important source of soluble nutrients and metals during stratification. Evidence for this was indicated by seasonal concentration increases of organic carbon, nitrogen, phosphorus, iron, and manganese in bottom waters. Data collected in September also indicated the occurrence of marked vertical variations for these variables. Concentrations were highest near the sediment/water interface, and gradients of decreasing values were apparent in the water column, suggesting the influence of sediment/water interactions.

Chemical dynamics at the sediment/water interface are often strongly associated with anaerobic conditions. For instance, iron and manganese are reduced to soluble forms under these conditions. Since soils inundated during impoundment of Russell Lake contained appreciable amounts of iron and manganese, release of reduced forms from the sediment under anoxic conditions was a potential concern. It appeared that increased levels of these variables in the bottom waters coincided with the occurrence of anoxia. That a major portion of these total concentrations was in the dissolved form in the hypolimnion further suggests reduction of iron and manganese.

Release of soluble phosphorus from sediments and inundated soils is also often associated with iron and manganese reduction. It has been well established that interactions between iron and phosphorus can result in the accumulation of soluble phosphorus in the hypolimnion during summer stratification (Mortimer, 1971; Larson et al., 1981; Kennedy et al., 1983; James and Kennedy, 1985). Russell Lake experienced a marked increase in soluble reactive phosphorus and soluble iron concentrations in the hypolimnion during the first year of impoundment. While the cause of this pattern in Russell Lake has not been established, iron phosphorus interactions at the sediment surface is one possible mechanism.

Phosphorus is an important nutrient for algal growth, and its dynamics and pattern of distribution often influence algal productivity. Using a radioactive tracer (32-P) to monitor phosphate uptake rates by planktonic communities, Heath (1985) reported marked differences in phosphate uptake along the longitudinal axis of Russell Lake. Uptake kinetics were slow above the

plunge point, indicative of a phosphate-sufficient environment for algae.
However, uptake rates increased markedly at the plunge point, suggesting that
phosphate becomes limiting in this region. Near the dam, phosphate availabil-
ity was extremely limited and uptake rates decreased. These results indicate
that the planktonic community in a majority of the main basin are growth-
limited by the availability of phosphate. Kimmel et al. (1985) observed
highest chlorophyll a concentrations below the plunge point and gradients of
decreasing values toward the dam during the summer months. These patterns
corresponded with Heath's observations of phosphate availability and uptake
kinetics, and have important implications for algal distribution patterns.
In subsequent years internal recycling from the more phosphorus-rich hypo-
limnion to the epilimnion may become an important source for algal growth
in areas that exhibit extensive hypolimnetic phosphorus accumulation (see
Kennedy et al., 1983).

Fall turnover was a critical event during the first year of impoundment.
The buildup of hypolimnetic organic carbon and reduced metals during anoxia
led to reduced dissolved oxygen concentrations in surface strata during mix-
ing. The progressive erosion of the anoxic zone led to increased levels of
iron and manganese in the upper water column, suggesting entrainment of read-
ily oxidizable nutrients and metals during epilimnetic expansion. During this
period dissolved oxygen concentrations decreased substantially in the epilim-
netic zone, remaining at levels of less than 3.0 mg/l in a majority of the
water column until complete turnover occurred. While water column mixing and
turnover can generally result in replenishment of dissolved oxygen to near-
saturation, it appeared that upward entrainment of oxidizable material (i.e.,
iron, manganese, organic carbon) resulted in a dissolved oxygen demand in the
upper water column that could not be met by reaeration.

Turnover was a temporal and spatial event, as suggested by the marked
variations in temperature and dissolved oxygen on a lakewide basis. Because
of the characteristics of a basin morphometry which broadened and deepened
from the headwater regions to the dam, complete mixing was first achieved
in the shallow upstream and embayment areas. Turnover in the forebay area
occurred later and lasted several months. This was due, in part, to the
large volume of stratified water which had to be mixed.

Interactions were identified between Hartwell Lake discharges and the
headwater region of Richard B. Russell Lake. During peak power generation,

Hartwell Lake discharges greatly modified the physical and chemical structure
of the upper reaches of Richard B. Russell Lake on a diel time cycle. This
was detected by a diel monitoring study conducted during a generation cycle.
In general, cool release waters moving at high inflow velocities resulted in
the displacement of lake water and the establishment of a near-riverine envi-
ronment from Station 200 to near Station 180. At night, however, upstream
currents were observed as displaced lake water moved back over the cooler
release water. These ebb and flow movements created a highly dynamic mixing
regime which may affect the exchange and movement of nutrients in the upper
end of Richard B. Russell Lake. These interactions have a strong bearing on
nutrient cycles along the lake's axis and the biological community in the
headwater region.

Apparent during the course of routine water quality monitoring and the
diel study were decreased dissolved oxygen concentrations in the upper regions
of Richard B. Russell Lake as a result of Hartwell Lake discharges. These de-
clines were seasonal and related to hypolimnetic dissolved oxygen depletion
in Hartwell Lake. During July through October, concentrations approaching
4.0 mg O_2/l were detected in the Richard B. Russell Lake headwaters following
generation at Hartwell Dam. These low levels are of potential concern to fish
habitats in this region. Further, the movement of these waters downstream may
further impact dissolved oxygen conditions, particularly in the region of the
oxygen injection system.

Hartwell inflows plunge in the vicinity of Station 180 (approximately
13 km downstream from Hartwell Dam) and continue down-reservoir as an inter-
flow, as indicated by a dye-tracing study. The depth of interflow was ap-
proximately 4 to 5 m, which has implications for the placement of dissolved
oxygen, nutrients, and metals from Hartwell Dam into Russell Lake.

Indirect evidence of the advective movement of Hartwell Lake discharges
was suggested with longitudinal patterns of chemical concentrations. These
data supported a contention that interflows may have been occurring at the
depth of the tainter gate withdrawal zone during summer stratification. For
instance, specific conductance data often displayed a longitudinal zone of low
values which extended from the dam to Hartwell Lake tailwaters. This stratum
of water was located below the epilimnion in the region of the withdrawal
zone. Concentrations of total and dissolved organic carbon and nitrogen,
total magnesium, calcium, sodium, and potassium, and hardness were low in

this zone and reflective of concentrations observed in the Hartwell Lake tailwater region. These data suggest the occurrence of flow patterns created by the interaction of Hartwell Lake discharges and tainter gate releases at Russell Dam.

Such movement of inflows through reservoirs is not uncommon and has been frequently observed in other reservoirs. For instance, Ford and Johnson (1981) observed the movement of stormwater inflows through DeGray Lake at the depth of the withdrawal zone during operational releases. Ebel and Koski (1968) reported the development of a metalimnetic dissolved oxygen minimum in Brownlee Reservoir as a result of movement of upstream anoxic water toward the middepth outlet structure. A similar metalimnetic dissolved oxygen pattern was attributed to in-pool flow patterns created by a metalimnetic discharge regime in DeGray Lake (Nix, 1981). Nix (1981) detected the advective transport of dissolved iron and manganese down-reservoir in the area of interflow during summer.

Additional evidence of the importance of advective transport in Richard B. Russell Lake was suggested by observations during the turnover period. While dissolved oxygen concentrations were depressed in a major vertical area of the forebay region, a progressive intrusion of water high in dissolved oxygen was detected between the 20- and 40-m depths from mid-November until late-December. This maximum in dissolved oxygen was first detected at Station 120 and later at Station 60. This pattern suggested the possible occurrence of inflows moving toward the dam as an underflowing density current.

The depth at which inflows move through Russell Lake may change when discharges occur through penstocks. This would have important implications for the water quality of the discharges to Clarks Hill Lake in subsequent years. Interflows occurring at the depth of the midhypolimnetic withdrawal zone may entrain anoxic water high in dissolved nutrients and metals from upstream locations (i.e., Stations 120 and 160). Input of these materials to the forebay region may impact the dissolved oxygen budget and the performance of the oxygen injection system. Therefore, future studies should address in-pool flow patterns and water quality characteristics at the depth of the withdrawal zone in the forebay area.

Clarks Hill Lake

Water quality conditions at Clarks Hill Lake differed considerably from

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those at Richard B. Russell Lake during the first year of study. Hypolimnetic dissolved oxygen depletion was less extensive in Clarks Hill Lake, and anoxic conditions were observed in the forebay area for only a short period of time. Hypolimnetic chemical patterns were moderate compared to Richard B. Russell Lake while the concentrations of soluble organic carbon, phosphorus, and iron did not exhibit appreciable increases during the stratified period; soluble manganese and nitrogen forms did display significant hypolimnetic increases.

Releases from Richard B. Russell Lake provided well-oxygenated water to the upper end of Clarks Hill Lake during 1984. Discharges of oxygenated water from near-surface tainter gates were sufficiently aerated to provide inflow concentrations greater than 8.0 mg/l to Clarks Hill Lake during the stratified period. Had releases occurred from sluice gates during this period, the release of anoxic water high in soluble metal and nutrient concentrations could have led to significant water quality problems in the upper reaches of Clarks Hill Lake.

Inflow dynamics were unique to Clarks Hill Lake due to the operation of tainter gates throughout the stratified period. Releases were generally continuous rather than pulsed which allowed for the establishment of near-steady state conditions with respect to inflow regime. Inflows created a riverine environment in the tailwater area, and a well-defined density current was established near Station 40. Evidence for this came from routine water quality observations and the diel study. The existence of density inflows was suggested by the occurrence of high levels of dissolved oxygen within middepth strata at Station 40. However, this inflow pattern is expected to change when on-line power generation occurs. Water quality conditions of release water may also change as a result of hypolimnetic withdrawal, depending on conditions in Richard B. Russell Lake next year and the performance of the oxygen injection system.

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PART VI: SUMMARY

Richard B. Russell Lake

Results of the water quality studies at Richard B. Russell Lake are sum-
marized below.

1. The impoundment of Richard B. Russell Lake has resulted in a signif-
   icant change in the hydrological characteristics of the impounded
   reach of the Savannah River and in the headwater area of Clarks Hill
   Lake.

2. Filling of Richard B. Russell Lake was initiated in December 1983,
   and full pool elevation was attained in November 1984. The most
   rapid pool elevation increases occurred in three phases from Decem-
   ber 1983 until April 1984. Phase I represented preimpoundment pool
   elevation, Phase II was marked by a 7.5-m pool rise, and Phase III
   represented a pool elevation increase from 110 to 140 m MSL from
   December until February. Limnological conditions during these
   phases of filling reflected changes in pool elevation, precipita-
   tion, hydraulic residence time, increases in surface area of in-
   undated bottomlands, and discharges.

3. Discharges from Richard B. Russell Lake occurred from bottom sluice
   gates during October 1983 through May 1984. Tainter gate releases
   were employed from May through December 1984, and penstock dis-
   charges occurred during December 1984.

4. During Phase I of filling, Richard B. Russell Lake exhibited water
   quality conditions that were characteristic of inflowing waters.
   Temperature was uniform in the water column, and dissolved oxygen
   concentration was high. Chemical concentrations were low and re-
   flective of concentrations of the inflowing waters.

5. Effects of a submerged coffer dike on flow patterns were noted dur-
   ing Phase II. The breached dike appeared to partially impound bot-
   tom waters, allowing for preferential discharges of surface waters
   from the pool.

6. The large increase in pool rise during Phase III resulted in
   greater influences from the Beaverdam Creek and Rocky River embay-
   ments. Specific conductance values often reflected the occurrence
   of inflows via secondary tributaries during storm events.

7. Turbidity, organic carbon, nitrogen forms, phosphorus, and metals
   often exhibited higher concentrations in the two major tributary
   embayments during Phase III, suggestive of the occurrence of load-
   ing via these tributaries.

8. Approximately 7,700 ha of bottomland, which were rich in organic
   material and readily solubilized nutrients and metals, were inun-
   dated during Phase III. Although it was anticipated that chemical
   concentrations might immediately increase in the water column due
   to release of material from soils, chemical changes were moderate.
   The existence of cool water temperatures, isothermal conditions,

and high dissolved oxygen concentrations might have influenced chemical conditions during this period.

9. The stabilization of pool elevation in March and warmer air temperatures led to stratified conditions, which lasted from March until September. Basin morphology and hydrological characteristics had an impact on thermal structure during this period. Pool elevation was maintained 1.5 m below power pool elevation, and releases were discharged through near-surface tainter gates.

10. Lakewide surface heating began in late March, and stratified conditions were established by May. Thermal stratification was evident in the two major embayments and in the main basin from Station 60 to Station 160. Upstream of Station 160, thermal patterns were disrupted during peaking power generation from Hartwell Dam. Inflows from Hartwell Dam appeared to plunge in the vicinity of Station 180.

11. Throughout much of the stratified period the epilimnion was shallow (i.e., 3-5 m thick) and apparently influenced by the near-surface discharge regime.

12. Hypolimnetic dissolved oxygen depletion began in the two major embayments shortly after the establishment of thermal stratification in March, spread to the confluence of the two tributaries (i.e., Station 120), then to the dam and Station 160 by June.

13. Throughout the stratified period, hypolimnetic anoxia progressed upward in the hypolimnetic water column throughout much of the main basin and the two tributary embayments.

14. Midhypolimnetic discharges from Hartwell Dam exhibited dissolved oxygen concentration near 5.0 mg/l from July until October and influenced dissolved oxygen concentrations in the headwater region of Richard B. Russell Lake.

15. pH exhibited moderate patterns and ranged from near 6 to 7 pH units during thermal stratification.

16. The oxidation-reduction potential exhibited negative values in areas where hypolimnetic anoxia occurred, indicative of the establishment of a reducing environment. The spatial development of negative redox potential coincided with the spatial development of anoxia.

17. Specific conductance displayed increases in the anoxic bottom waters which coincided with increased concentrations of nutrients and metals. Vertical gradients of increasing values toward the lake bottom were suggestive of interactions between the inundated bottomlands and the water column.

18. Specific conductance data also reflected the occurrence of density inflows originating from Hartwell Dam during stratification. Values were low and uniform in the thermocline region of Russell Lake and comparable to values observed in the Hartwell Dam tailwater area. In addition, Russell discharges were via tainter gates during summer stratification, suggesting advective movement of inflows toward the withdrawal zone. Values were higher in the epilimnion

and hypolimnion. These patterns suggest that Hartwell Lake inflows moved through Richard B. Russell Lake as an interflow.

19. Many of the chemical variables displayed seasonal and spatial concentration increases in the hypolimnion during the stratified period which coincided with similar patterns in the development of thermal stratification and dissolved oxygen depletion. Seasonally, hypolimnetic chemical concentrations began increasing first at the shallow embayment areas, then the main basin.

20. By September, hypolimnetic levels of these variables were elevated from Station 60 to Station 160, and at Stations 130, 140, and 150. Vertical concentration gradients were evident in these areas, suggesting the movement of nutrients and metals out of inundated bottomland and sediment into the water column. Bottom concentrations were generally highest in the forebay area and lower at Station 160. A major percentage of these chemical concentrations was in the soluble form.

21. Hartwell Lake discharges had a beneficial impact on chemical concentrations in the upstream areas of Richard B. Russell Lake. Concentrations were generally low in this region.

22. Like specific conductance, a band of uniform nutrient concentrations was observed in the metalimnetic area which stretched from Russell Dam to the Hartwell Dam tailwater. Concentrations were generally low in this region and comparable to concentrations observed in the tailwaters. This suggests the movement of Hartwell Dam inflows through Richard B. Russell Lake as an interflow. Exemplary of this effect were patterns in the concentration of magnesium, calcium, sodium, and potassium.

23. Nitrate-nitrite nitrogen also exhibited increased levels within this stratum. Above and below this zone, concentrations were undetectable.

24. Concentrations of total and dissolved chemical forms were generally low in the surface waters throughout the stratified period. Slightly elevated seasonal peaks were often related to influences from Hartwell Dam and to increased algal productivity.

25. Chlorophyll a exhibited a lakewide concentration increase from July until August. Concentration was generally highest at Station 180, and lower, more uniform values occurred toward the dam. Above Station 180, concentration was low and appeared to be influenced by Hartwell Dam releases.

26. Seasonally cooler air temperatures led to surface cooling, epilimnetic expansion, and turnover during August through December. Isothermal conditions were observed in the two embayment areas in mid-November and in the forebay area by mid-December.

27. Epilimnetic expansion and erosion of the anoxic zone resulted in severely depressed dissolved oxygen levels in the upper water column in much of the reservoir.

28. The depression of dissolved oxygen during turnover was attributed to exchanges between the epilimnion and hypolimnion. The

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redistribution of anoxic water containing readily oxidizable materials created a demand on dissolved oxygen stores which could not be immediately met by reaeration for several weeks.

29. The recovery to higher concentrations of dissolved oxygen concentrations (>6.0 ppm) varied spatially. Upstream main-basin and embayment areas exhibited recovery by 29 November, while recovery in the forebay area did not occur until late December.

30. Also apparent during the turnover period was the progressive intrusion of water high in dissolved oxygen at near-bottom depths. This was apparently related to the occurrence of density underflows.

31. Total and dissolved forms of iron and manganese increased in concentration in the upper water column during mixing. Complete turnover and reintroduction of dissolved oxygen to the entire water column resulted in an eventual reduction in these concentrations.

32. Releases occurred through tainter gates from August to early November. Releases were switched to a combination tainter gate and penstock operation during generator unit testing from November through December. Because the water quality of the discharges was poor, releases were stopped during a major portion of November, resulting in a pool elevation rise to 145 m MSL.

Clarks Hill Lake

Results of the water quality studies at Clarks Hill Lake are summarized below.

1. Water quality studies were initiated on Clarks Hill Lake in October 1983. The first year of data collection provided baseline information on water quality characteristics prior to penstock releases from Richard B. Russell Lake and implementation of the oxygen injection system.

2. Thermally stratified conditions were established at Station 20 from May until September, and mixing and turnover occurred from October until November. Epilimnetic thickness varied from 3-5 m in June to greater than 10 m by September.

3. At Station 40, stratified conditions were observed from April until September; however, epilimnetic thickness did not vary. The movement of Richard B. Russell Lake inflows as a density current below the thermocline may have influenced this pattern. Turnover at Station 40 occurred in November.

4. Dissolved oxygen patterns varied longitudinally in the hypolimnion during the stratified period. Apparent at Station 20 was gradual hypolimnetic dissolved oxygen depletion from June until August, and anoxia from September until early November. Anoxic conditions were observed from the lake bottom to the 25-m depth in mid-October. Turnover resulted in a rapid dissolved oxygen recovery to >6.0 mg/l.

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5. At Station 40, hypolimnetic dissolved oxygen concentrations remained above 6.0 mg/l throughout the study period. The movement of a well-oxygenated density current through this region directly influenced dissolved oxygen patterns.

6. Longitudinal patterns in temperature, dissolved oxygen, and specific conductance supported the contention that cool, well-oxygenated inflows originating from tainter gate releases entered Clarks Hill Lake as a density interflow.

7. Spatial and seasonal patterns for many chemical variables were moderate. Total organic carbon, total phosphorus, and total iron exhibited slightly higher concentrations at Station 40 during the winter mixed period, but between-station differences were minimal during summer stratification. Concentration increases were not detected for these variables in the anoxic hypolimnion during summer stratification at Station 20.

8. Other variables displayed concentration increases in the hypolimnion at Station 20 during summer stratification. Hypolimnetic concentrations of total nitrogen and dissolved nitrogen increased at this station from May until September. Total manganese and dissolved manganese displayed marked increases from July until September which were associated with the establishment of anoxia.

9. Longitudinal differences were observed between patterns for iron and manganese. Highest concentrations of iron were observed at Station 40 while manganese concentrations were highest at hypolimnetic depths at Station 20. These differences may be attributed to influences of Richard B. Russell Lake releases on patterns at Station 40. Differences between iron and manganese at Station 20 may be related to the fact that manganese is reduced to soluble forms at higher oxidation-reduction potentials than is iron and therefore accumulates in the hypolimnion before iron does.

PART VII: RECOMMENDATIONS

Major concerns during the first year of impoundment were potential
changes in water quality conditions in Richard B. Russell Lake and water qual-
ity effects in Clarks Hill Lake. Richard B. Russell Lake exhibited a large
zone of hypolimnetic anoxia, and the accumulation of soluble nutrients and
metals in the bottom waters during summer stratification. However, there was
minimal impact on water quality conditions in Clarks Hill Lake in 1984 as a
result of impoundment of Richard B. Russell Lake. This was primarily due to
the discharge of surface waters from Russell Dam via tainter gates. Since im-
paired water quality conditions are expected to continue in Richard B. Russell
Lake, operation and efficiency of the oxygen injection system will be critical
with respect to the quality of the discharges when hypolimnetic withdrawal oc-
curs during on-line production. Therefore, several recommendations for limno-
logical studies are offered.

Sampling efforts and special studies of the performance of the oxygen
injection system in Richard B. Russell Lake and the quality of the discharges
should be conducted. Intensive field sampling in the forebay region would
characterize: (1) the extent and duration of anoxia during summer stratifi-
cation, and (2) vertical patterns of oxygen in the water column during oxygen
injection.

Important in the efficient operation of the oxygen injection system
will be the estimation of the dissolved oxygen demand in Richard B. Russell
Lake. Therefore, a study of hypolimnetic mixing and in-pool flow patterns
should be conducted to gain a better understanding of the movement of water
in the area of oxygen injection and in the withdrawal zone. Continuous mon-
itoring of water quality conditions should also be conducted in the tail-
waters of Richard B. Russell Lake to identify impacts of discharges on
Clarks Hill Lake.

Routine water quality monitoring (i.e., chemical and in-situ sampling)
should continue on both lakes at selected stations. These studies will pro-
vide further information on water quality patterns in Richard B. Russell
Lake following impoundment and impacts on Clarks Hill Lake. Additional sta-
tions should be located in Clarks Hill Lake to more clearly define longitudi-
nal patterns in water quality which may occur due to releases from Russell
Dam. Since the Broad River may also have a significant impact on Clarks

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Hill Lake, a sampling station should be located in this region.

Intensive sampling of iron and manganese should be conducted at selected stations in Richard B. Russell and Clarks Hill Lakes throughout the stratified period and during turnover. It was inferred from results in 1984 that soluble metals can have a marked impact on dissolved oxygen dynamics in Richard B. Russell Lake. Since iron and manganese readily oxidize in an oxic environment, these materials may exert a significant demand on dissolved oxygen stores. This has a bearing on the operation and performance of the oxygen injection system and on the quality of the discharges.

Finally, special studies should be conducted in the upper end of Richard B. Russell and Clarks Hill Lakes to determine inflow patterns during generation cycles. It will be important to identify spatial and temporal trends in the headwater region of each lake, as well as the placement of inflows in the water column.

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Table 1

Physical Characteristics for Clarks Hill,
Richard B. Russell, and Hartwell Lakes

Characteristics	Clarks Hill Lake	Richard B. Russell Lake	Hartwell Lake
Maximum power pool elevation, m MSL	100.6	144.8	201.2
Minimum power pool elevation, m MSL	95.1	143.2	190.5
Flood control pool elevation, m MSL	102.1	146.3	202.7
Surface area, sq km*	283.2	107.9	226.6
Volume, 10^6 cu m*	3,023.3	1,271.0	3,146.7
Maximum depth, m*	45	47	55
Mean depth, m*	11	12	14
Reservoir length, km*	63	45	79 Tugaloo R. 72 Seneca R.
Shoreline length, km*	1,930	885	1,548
Shoreline development ratio*	32.0	24.5	29.0
Drainage area, sq km	15,906	7,508	5,406
Residence time, day*	144	102	306
Mean discharge, cms	243.6**	143.8†	119.0**
Top of penstock elevation, m MSL		133.2	
Bottom of penstock elevation, m MSL		110	

*   At maximum power pool elevation.
** Mean discharge for the period 1925-1973.
†   Mean river discharge for 31 yr.

Table 2

Water Quality Characteristics of Clarks Hill,
Richard B. Russell, and Hartwell Lakes*

Variable	Clarks Hill Lake**	Richard B. Russell Lake†	Hartwell Lake††
Dissolved oxygen	8.1	7.0	9.4
Temperature, deg. C	19.9	16.5	16.3
Specific conductance, umhos/cm	78.3	34.4	34.3
Turbidity	4.7	4.1	5.0
Total alkalinity	15.7	10.1	6.0
Total organic carbon	3.3	2.3	2.0
Total phosphorus	0.03	0.0	0.02
Soluble reactive phosphorus	0.02	0.0	0.01
Total nitrogen	--	0.42	--
Dissolved nitrogen	--	0.36	--
TKN	0.65	--	0.37
Ammonia-nitrogen	0.08	0.0	0.08
Nitrate-nitrite nitrogen	0.24	0.01	0.10
Total manganese	0.04	0.02	0.08
Dissolved manganese	0.05	0.18	0.12
Total iron	0.75	0.42	0.14
Dissolved iron	0.04	0.13	0.01
pH	6.7	6.6	6.4
Total solids	--	27.4	--
Suspended solids	2.2	3.0	2.5
Dissolved solids	24.2	--	44.5
Sulfate	2.0	3.5	1.1
Chloride	1.8	1.7	1.7

* Values represent annual means from surface waters near the dam.
** Clarks Hill Lake Water Quality Study (US Army Corps of Engineers, 1982a).
† Richard B. Russell Lake (this report).
†† Hartwell Lake Water Quality Study (US Army Corps of Engineers, 1982b):

Table 3

Sampling Schedule

Parameter	FY 84 Post Mixing			Winter		Spring High Flow			Stratification Early		Stratification Late		FY 85 Post Mixing		
	Oct	Nov	Dec	Jan	Feb	Mar	Apr	May	Jun	Jul	Aug	Sep	Oct	Nov	Dec

Scheduled Sampling (per Scope of Work)

Parameter	Oct	Nov	Dec	Jan	Feb	Mar	Apr	May	Jun	Jul	Aug	Sep	Oct	Nov	Dec
In situ	X	X	X	X	X	X	X	X	X	X	X	X	X	X	X
Physicochemical				X*		X	X	X	X	X		X			
Biological	X			X*		X	X	X	X			X			
Intensive studies											X				

Unscheduled Sampling

Parameter	Oct	Nov	Dec	Jan	Feb	Mar	Apr	May	Jun	Jul	Aug	Sep	Oct	Nov	Dec
In situ	X	X	X		X	X	X	X	X	X	X	X	X	X	X
Physicochemical		X			X	X	X	X		X			X	X	X
Biological	X	X	X		X	X	X	X	X		X		X	X	X

* Postmixing and winter combined.

Table 4

List of Variables

In situ

    Temperature
    Dissolved oxygen
    pH
    Specific conductance
    Oxidation-reduction potential

Physicochemical

    Turbidity
    Color
    Total solids
    Total nonfilterable solids
    (Suspended solids)
    Total alkalinity
    Chloride

Metals

    Total iron
    Dissolved iron
    Total manganese
    Dissolved manganese
    Total calcium
    Total potassium
    Total sodium
    Total magnesium

Nutrients

    Total carbon
    Total organic carbon
    Dissolved organic carbon
    Total phosphorus
    Total soluble phosphorus
    Soluble reactive phosphorus
    Total nitrogen
    Total soluble nitrogen
    Ammonium-nitrogen
    Nitrate-nitrite nitrogen
    Sulfate
    Sulfide

Additional Metals

    Arsenic
    Cadmium
    Chromium
    Copper
    Lead
    Mercury
    Nickel
    Zinc

Biological

    Chlorophyll a

Table 5

Maximum Volumetric Depletion Rates (mg/l/day)

for Richard B. Russell Lake in 1984

Based on PROFILE Calculations

Station	Date	Hypolimnion	Metalimnion	Both
60B	11 Jun-25 Jun	0.09	0.11	0.11
120	22 Mar-2 Apr	0.22	0.20	0.21
130	6 Mar-19 Mar	0.68	0.46	0.51
140	6 Mar-19 Mar	0.59	0.20	0.29
160	14 May-29 May	0.36	0.13	0.21

Notes: Top elevation of hypolimnion: 125 m.
       Top elevation of metalimnion: 135 m.

Table 6

Maximum Volumetric Depletion Rates by Layer

for Richard B. Russell Lake in 1984

Elevation m	First Date	First DO mg/l	Second Date	Second DO mg/l	Depletion Rate mg/l per day
			Station 60		
143.4	25 Jun	7.7	30 Jul	7.6	0.00
141.6	11 Jun	8.5	25 Jun	7.7	0.06
139.6	25 Jun	8.9	30 Jul	4.5	0.13
137.6	19 Mar	11.0	02 Apr	9.6	0.10
135.6	30 Apr	8.4	14 May	6.6	0.13
133.6	30 Apr	8.3	14 May	6.6	0.12
131.6	11 Jun	5.5	25 Jun	3.6	0.14
129.6	11 Jun	5.5	25 Jun	3.7	0.13
127.6	11 Jun	5.7	25 Jun	4.0	0.12
125.6	25 Jun	4.3	30 Jul	0.0	0.12
123.6	25 Jun	4.5	30 Jul	0.0	0.13
121.6	19 Mar	9.4	02 Apr	7.4	0.14
119.6	19 Mar	9.4	02 Apr	7.0	0.17
117.6	19 Mar	9.2	02 Apr	6.8	0.17
115.6	19 Mar	9.0	02 Apr	6.5	0.18
113.6	11 Jun	3.9	25 Jun	0.4	0.25
111.6	29 May	4.2	11 Jun	1.6	0.20
109.6	29 May	3.1	11 Jun	0.3	0.22
107.6	19 Mar	8.0	02 Apr	6.0	0.15
105.6	19 Mar	7.8	02 Apr	6.0	0.13
103.6	19 Mar	7.6	02 Apr	5.9	0.12
101.6	19 Mar	7.3	02 Apr	5.8	0.11
			Station 120		
143.3	25 Jun	6.0	11 Jul	9.9	-0.24
141.6	14 May	9.8	29 May	6.7	0.21
139.6	22 Mar	9.3	02 Apr	7.4	0.17
137.6	14 May	7.7	29 May	4.7	0.20
135.6	22 Mar	9.3	02 Apr	7.4	0.17
133.6	22 Mar	9.3	02 Apr	7.2	0.19
131.6	22 Mar	9.3	02 Apr	7.0	0.21
129.6	22 Mar	9.3	02 Apr	7.0	0.21
127.6	22 Mar	9.2	02 Apr	6.9	0.21
125.6	14 May	8.2	29 May	5.4	0.19
123.6	14 May	8.1	29 May	5.3	0.19
121.6	14 May	8.1	29 May	5.2	0.19
119.6	22 Mar	7.6	02 Apr	5.5	0.19
117.6	22 Mar	6.9	02 Apr	4.3	0.23
115.6	22 Mar	6.5	02 Apr	3.2	0.30

(Continued)

Table 6 (Continued)

Elevation m	First Date	First DO mg/l	Second Date	Second DO mg/l	Depletion Rate mg/l per day
			Station 120 (Continued)		
113.6	22 Mar	5.8	02 Apr	2.6	0.30
111.6	22 Mar	5.0	02 Apr	1.7	0.30
109.6	22 Mar	4.8	02 Apr	1.7	0.28
107.6	22 Mar	4.8	02 Apr	1.7	0.28
105.6	22 Mar	4.8	02 Apr	1.7	0.28
103.6	22 Mar	4.8	02 Apr	1.7	0.28
101.6	22 Mar	4.8	02 Apr	1.7	0.28
			Station 130		
141.8	14 May	7.7	29 May	5.9	0.12
139.8	14 May	5.4	29 May	2.1	0.22
137.8	22 Mar	8.7	02 Apr	5.6	0.28
135.8	22 Mar	8.4	02 Apr	4.5	0.35
133.8	22 Mar	6.9	02 Apr	2.3	0.42
131.8	06 Mar	8.9	19 Mar	3.4	0.42
129.8	06 Mar	8.9	19 Mar	1.5	0.57
127.8	06 Mar	8.8	19 Mar	0.0	0.68
125.8	06 Mar	8.8	19 Mar	0.0	0.68
123.8	06 Mar	8.8	19 Mar	0.0	0.68
121.8	06 Mar	8.8	19 Mar	0.0	0.68
119.8	06 Mar	8.8	19 Mar	0.0	0.68
117.8	06 Mar	8.8	19 Mar	0.0	0.68
115.8	06 Mar	8.8	19 Mar	0.0	0.68
113.8	06 Mar	8.8	19 Mar	0.0	0.68
111.8	06 Mar	8.8	19 Mar	0.0	0.68
109.8	06 Mar	8.8	19 Mar	0.0	0.68
			Station 140		
141.8	14 May	10.2	29 May	7.7	0.17
139.8	29 May	7.4	11 Jun	5.4	0.16
137.8	14 May	6.1	29 May	0.2	0.39
135.8	14 May	4.0	29 May	0.1	0.26
133.8	22 Mar	8.2	02 Apr	6.3	0.17
131.8	30 Apr	4.1	14 May	1.4	0.19
129.8	22 Mar	5.4	02 Apr	2.9	0.23
127.8	06 Mar	7.9	19 Mar	4.2	0.29
125.8	06 Mar	7.9	19 Mar	0.8	0.55
123.8	06 Mar	8.0	19 Mar	0.6	0.57

(Continued)

Elevation m	First Date	First DO mg/l	Second Date	Second DO mg/l	Depletion Rate mg/l per day
		Station 140 (Continued)			
121.8	06 Mar	8.0	19 Mar	0.1	0.61
119.8	06 Mar	8.0	19 Mar	0.1	0.61
117.8	06 Mar	8.0	19 Mar	0.1	0.61
115.8	06 Mar	8.0	19 Mar	0.1	0.61
113.8	06 Mar	8.0	19 Mar	0.1	0.61
111.8	06 Mar	8.0	19 Mar	0.1	0.61
109.8	06 Mar	8.0	19 Mar	0.1	0.61
		Station 160			
143.4	25 Jun	6.4	11 Jul	9.2	-0.18
141.6	14 May	8.4	29 May	6.9	0.10
139.6	19 Mar	11.1	02 Apr	8.3	0.20
137.6	19 Mar	10.8	02 Apr	8.0	0.20
135.6	19 Mar	10.3	02 Apr	7.6	0.19
133.6	19 Mar	9.9	02 Apr	7.3	0.18
131.6	19 Mar	9.3	02 Apr	7.3	0.15
129.6	11 Jun	5.5	25 Jun	2.9	0.18
127.6	14 May	7.4	29 May	4.8	0.18
125.6	14 May	7.4	29 May	4.0	0.23
123.6	14 May	7.2	29 May	3.0	0.28
121.6	14 May	6.5	29 May	0.8	0.38
119.6	14 May	6.4	29 May	0.6	0.39
117.6	14 May	6.4	29 May	0.6	0.39
115.6	14 May	6.4	29 May	0.6	0.39
113.6	14 May	6.4	29 May	0.6	0.39
111.6	14 May	6.4	29 May	0.6	0.39
109.6	14 May	6.4	29 May	0.6	0.39
107.6	14 May	6.4	29 May	0.6	0.39
105.6	14 May	6.4	29 May	0.6	0.39
103.6	14 May	6.4	29 May	0.6	0.39
101.6	14 May	6.4	29 May	0.6	0.39

Table 7

Maximum Areal Depletion Rates (mg/sq m/day)

for Richard B. Russell Lake in 1984

Based on PROFILE Calculations

Station	Date	Hypolimnion	Metalimnion	Both
60B	30 Apr-14 May	221	587	678
120	22 Mar-2 Apr	2,191	1,318	2,218
130	6 Mar-19 Mar	3,953	2,935	4,286
140	6 Mar-19 Mar	3,464	1,254	2,438
160	14 May-29 May	3,537	848	2,300

Notes: Top elevation of hypolimnion: 125 m.
       Top elevation of metalimnion: 135 m.

Table 8

Maximum Areal Depletion Rates (mg/sq m/day)

for Richard B. Russell Lake in 1984

Based on PROFILE Calculations

Station	Date	Hypolimnion	Metalimnion	Both
020	31 May-18 Jun	1,080	965	1,399
030	31 May-18 Jun	3,185	179	1,458
040	1 May-31 May	1,755	567	1,272

Notes: Top elevation of hypolimnion: 85 m.
       Top elevation of metalimnion: 95 m.

Table 9

Dates, Times (hr), and Locations of Diel

Sampling at Clarks Hill Lake

	Sampling Round					
Station	1 (13 Aug)	2 (13 Aug)	3 (13 Aug)	4 (13 Aug)	5 (14 Aug)	6 (15 Aug)
40	0900	1155	1440	2200	1120	1300
42	0930	1130	1410	2130	1100	1245
45	0813	1045	1350	2100	1035	1230
46	0800	1030	1330	2040	1020	1220
47	--	--	--	--	0955	1205
48	0745	1015	1315	2030	0940	1200
49	--	--	--	--	--	1150
50	0725	1000	1300	2020	0930	1145

Table 10

Dates, Times (hr), and Locations of Diel

Sampling at Richard B. Russell Lake

Station	Sampling Round								
	1 (13 Aug)	2 (13 Aug)	3 (13 Aug)	4 (13 Aug)	5 (14 Aug)	6 (14 Aug)	7 (14 Aug)	8 (14 Aug)	9 (14 Aug)
163	--	--	--	--	0030	0615	--	--	--
164	0748	1137	1514	2100	0100	0435	0820	1130	1550
166	0816	1205	1541	2200	0135	0505	0845	1200	1610
167	0849	1227	1559	2235	0200	0530	0910	1215	1630
180	0905	1310	1640	2350	--	0515	0956	1248	1642
182	0840	1250	1620	2210	0020	--	0937	1233	1639
184	0825	1235	1600	2050	0130	0340	0923	1221	1615
187	0810	1215	1545	2120	0050	0400	0911	1200	1607
190	0800	1200	1535	2140	0110	0420	0857	1152	1600
193	0750	1155	1530	--	--	--	0847	1141	1552

APPENDIX A: ANALYTICAL PROCEDURES

Sample Handling

A variety of sample containers were employed for transporting and/or storing field samples. Acid-washed, 1-1, linear polyethylene (LPE) bottles were used for samples for analyses of nutrients, metals, color, and chloride. LPE bottles (500-ml) were used for storing samples for solids analyses. Alkalinity and sulfate samples were contained in 50-ml LPE bottles, and care was taken to minimize shaking of the sample prior to analysis. A 60-ml brown LPE bottle was used for collection of turbidity samples. Acid-washed 60-ml LPE bottles and 50-ml syringes were used for anoxic samples collected using methods described below.

Field treatment of water samples included preservation, filtration, and the use of special collection techniques for anoxic samples. Preservation techniques involved the addition of nitric acid (to pH <2.0) to samples for metal analyses. Anoxic samples were filtered in the field via a manifold system or y-connector fitted with filter holders. Flow was reduced to avoid aeration of the incoming water, and the apparatus was flushed several times prior to filtration. Filters were washed in the field with sample water prior to collection of the filtrate. Sulfide samples were collected in a syringe in a similar manner and then dispensed into a basic solution to minimize loss. With the exception of samples for turbidity and those preserved with acid, all samples were stored in the dark and on ice until processed in the laboratory. Anoxic samples were not analyzed for turbidity, color, and nonfilterable solids.

Laboratory Protocol and Quality Control

Laboratory water treatment. Water from a well located onsite was processed by deionization and distillation, reverse osmosis (RO), or RO followed by ultrapurification (Nanopure, Barnstead, Boston, Mass.) involving organic removal, mixed-bed deionization, and filtration. Prior to distillation in an all-glass, 3-1/hr still (Corning "ACS," Glass Still, Corning Glass Company, Corning, N. Y.), the source water was deionized by a mixed-bed deionizer cartridge (Barnstead High Capacity). A 12-gal glass bottle was used for

collection, and all connections between the still and the bottle were glass.

The RO unit was a 40-l/hr Barnstead R.O. (Barnstead, Mass.) equipped with a 100-l plastic reservoir. Product water from this unit was available for general lab use and as the source water for the Nanopure system. This latter system employed a carbon removal cartridge (activated carbon), two mixed-resin bed cartridges (Barnstead Ultrapure), and a 0.2-μ filter cartridge. Processed water normally had a carbon concentration of less than 1 mg C/l and a resistance of approximately 3 megaohms. Routine monitoring of all systems was conducted to ensure product water quality.

Glassware preparation. In general, glassware and sample bottles were washed in soap and water with a brush, rinsed with hot tap water, rinsed three times with RO water, and allowed to air dry. Sample bottles and all glassware associated with nutrient and metal analyses were acid-washed in a dilute (10 percent) hydrochloric acid solution followed by five to seven rinses.

Reagents and chemicals. All reagents and chemicals used were of certified grade. Standards used were primary standards or were traceable to NBS standards. Analytical techniques (i.e., the use of analytical balances and certified volumetric glassware) were employed whenever necessary or specified for a particular analytical method.

Quality control. Analytical quality control was maintained by four techniques: standard calibration curves, split and replicate sampling, analysis of spiked samples, and analysis of EPA standards. Standards or standard curves were used for instrument calibration. Split and replicate samples were used to check sampling and analytical precision. Analysis of EPA standards and laboratory spiked samples provided a check of laboratory analytical accuracy.

Primary standards were used to calibrate both field and laboratory equipment, giving direct measurements of their respective parameters (e.g., conductivity, turbidity, pH). When the value of a parameter was determined by chemical reaction and/or color development (e.g., nutrients), standard curves covering the expected sample concentration range were established for each batch of samples. Standard curves were evaluated using appropriate statistical procedures, and the percent recovery of each standard was calculated.

While EPA recommends a new standard curve be run for each new set of reagents, such curves were run before each batch of samples. In the case of nutrients and carbon, additional standards were analyzed at the end of each

A2

batch. The slope, intercept, and correlation coefficient, as well as percent recovery of each standard, were subjectively compared to previous values. If the curve was suspect based upon this comparison, the run was rejected and the analyses were repeated.

Replicate samples were obtained from four randomly selected sampling locations during each sampling trip. These samples, which generally represented approximately 10 percent of the total number of samples, provided a means for estimating errors due to sampling and intrinsic variability. Coefficients of variation (CV) have been calculated for each variable for each replicate, and then a mean of the CV's has been calculated. These means (Table A1) represent the relative sampling precision and provide a method for comparing different analytical procedures.

The analytical precision of each assay was evaluated by splitting samples in the laboratory and analyzing each subsample separately. As with replicates, split samples were randomly selected for each sampling period. These samples provided a test of analytical reliability and a measure of the normal variability due to analysis. The CV was calculated for each split that had values above the detection limit, and mean CV's were calculated for each variable (Table A1).

The accuracy, or description of how closely analyzed values are to the actual values, was evaluated by analysis of EPA Quality Control Standards (Quality Assurance Branch, Environmental Monitoring and Support Laboratory, EPA, Cincinnati, Ohio) and by the analysis of unknown spiked samples prepared in the laboratory. Laboratory values were compared to actual EPA or spike values and recorded as percent or relative error (i.e., the difference between the actual and lab values expressed as a percent of the actual value). These results are presented in Table A2.

Table A1

Mean Coefficients of Variation for

Replicate and Split Samples

Variable	Replicate Samples	Split Samples
Dissolved oxygen	4.2	*
Specific conductance	1.8	*
pH	1.5	*
Turbidity	3.6	0.8
Sulfate	7.5	9.4
Total solids	13.6	13.2
Suspended solids	28.5	19.1
Total alkalinity	3.0	1.7
Total organic carbon	3.3	2.8
Dissolved organic carbon	5.0	4.5
Chloride	4.1	1.4
Total phosphorus	9.1	4.5
Total soluble phosphorus	10.4	12.5
Soluble reactive phosphorus	4.6	14.8
Total nitrogen	9.6	11.0
Total dissolved nitrogen	15.1	9.3
Ammonium-nitrogen	6.6	4.6
Nitrate-nitrite nitrogen	5.6	1.6
Total iron	10.9	10.4
Soluble iron	8.7	0.7
Total manganese	4.6	0.6
Soluble manganese	2.4	0.0
Total sodium	4.2	2.6
Total potassium	3.7	2.8
Total calcium	2.4	4.0
Total magnesium	2.6	1.3

* Missing data.

Table A2

Mean Percent Recovery for EPA Quality Control

Samples and Laboratory Spiked Samples

Variable	EPA Sample	Spike Samples
Total solids	98.0 ± 5.1	104.8 ± 16.9
Suspended solids	92.6 ± 2.8	*
Chloride	112.2 ± 6.0	104.5 ± 26.1
Turbidity	102.2 ± 8.8	*
Total alkalinity	95.2 ± 2.1	89.7 ± 10.0
pH	97.2 ± 1.5	*
Total organic carbon	93.4 ± 4.7	93.8 ± 18.7
Sulfate	*	95.5 ± 19.6
Total phosphorus	95.8 ± 1.5	96.0 ± 9.5
Total soluble phosphorus	*	90.6 ± 12.7
Soluble reactive phosphorus	105.4 ± 0.6	86.5 ± 8.5
Total nitrogen	*	102.3 ± 8.3
Total soluble nitrogen	*	97.2 ± 12.6
Ammonium-nitrogen	103.8 ± 44.5	96.7 ± 15.2
Nitrate-nitrite nitrogen	92.2 ± 12.6	89.8 ± 13.9
Total iron	106.2**	93.5 ± 11.4
Dissolved iron	*	97.8 ± 10.4
Total manganese	100.0**	88.3 ± 19.1
Dissolved manganese	*	99.1 ± 21.4
Total sodium	93.8 ± 1.8	106.0 ± 16.8
Total magnesium	*	89.3 ± 21.6
Total calcium	92.2 ± 7.6	89.1 ± 36.6
Total potassium	99.5 ± 0.7	108.6 ± 22.8

* Missing data.
** N = 1 .

Analytical Methods

Water Column Depth

Method: Depth Sounding
Detection Limit: 0.1 m

Secchi Disc Transparency

Method: Determination of depth of disappearance and reappearance of disc
Detection Limit: 0.1 m
Equipment: 20-cm Secchi disc with alternating black and white quadrats

Water Temperature*

Method: Thermistor thermometer
Detection Limit: 0.1 deg. C
Calibration: National Bureau of Standards certified thermometer

Dissolved Oxygen*

Method: Membrane electrode
Detection Limit: 0.1 mg/l
Calibration: Winkler titration on water in calibration tank. The residual
  current at zero dissolved oxygen concentration was determined for each
  probe using water deoxygenated by either (1) purging with N_2 or (2) satu-
  ration with sulfite.
Reference: APHA, 1980

Specific Conductance*

Method: Electrometric
Detection Limit: 1 umhos/cm
Calibration: Determination of specific conductance in calibration tank
  with conductivity bridge (Barnstead Model PM-70CB)
Reference: APHA, 1980
Comments: All readings were corrected for temperature to 25 deg. C

pH*

Method: Electrometric
Detection Limit: 0.1 pH unit
Calibration: Determination of pH in calibration tank with pH meter (Beckman
  Model Zeromatic IV) standardized with pH 7 and pH 4 buffer solutions

Oxidation-Reduction Potential*

Method: Electrometric
Calibration: Ferric/ferrous iron solution standardized to 475 mV
Reference: AHPA, 1980

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* In-situ measurements made with Hydrolab Surveyor or Martek Mark VIII
   equipment.

Turbidity

Method: Nephelometric
Detection Limit: 0.1 NTU
Calibration: Formazin standards per manufacturer's guidelines
Equipment: Hach Model 2100A Turbidimeter (Hach Corp., Ames, Iowa)
Reference: APHA, 1980
Sample Handling: Samples stored in the dark at ambient temperatures.
  Analyses conducted at room temperature within 24 hr of sample collection.
  Samples collected from anoxic water were not analyzed for turbidity.

Alkalinity

Method: Potentiometric titration
Detection Limit: 1.0 mg/l as $CaCO_3$
Calibration: pH meter standardized with pH 7 and pH 4 buffer solution
Equipment: pH meter; Beckman Model Zeromatic IV (Beckman Instruments);
  25-ml buret
Reference: APHA, 1980
Sample Handling: Analyzed within 24 hr of sample collection

Color

Method: Visual comparison
Detection Limit: 5 color units
Calibration: Serial dilution of platinum/cobalt standard for standard curve
Equipment: 50-ml matched Nessler tubes
References: APHA, 1980
Sample Handling: Analyzed within 24 hr of sample collection.
  Analyses performed after filtration through 0.45-µ membrane filter.
  Analyses were not performed on anoxic samples.

Solids

1. Total Solids (TS)
    Method: Total residue at 105 deg. C

2. Total Nonfilterable Solids (Suspended Solids) (SS)
    Method: Total nonfilterable residue retained by a 0.45-µ membrane
      filter at 105 deg. C
    Detection Limit: 0.1 mg/l
    Calibration: Analytical balance calibrated per manufacturer's
      guidelines with National Bureau of Standards approved weights
    Equipment: Analytical balance (Mettler, Model AC100, Mettler
      Instrument Corp., Hightstown, N. J.); drying oven (Fisher
      Model 501, Fisher Scientific, Pittsburgh, Pa.)
    Reference: APHA, 1980
    Sampling Handling: Stored at 4 deg. C prior to analyses. Total
      solids analyzed within 48 hr and total nonfilterable solids
      analyzed within 72 hr of sample collection.
    Comments: Filterable residue (dissolved solids) calculated as
      difference between TS and SS. SS analyses were not determined
      on anoxic samples.

Carbon

1. Total Carbon (TC)
   Method: Direct injection; infrared analysis

2. Total Organic Carbon (TOC)
   Method: Persulfate wet chemical oxidation (ampulated digestion);
     infrared analysis

3. Total Filterable Organic Carbon (DOC)
   Method: Persulfate wet chemical oxidation (ampulated digestion)
     on sample filtered through a glass fiber filter; infrared
     analysis
   Detection Limits: 0.2 mg/l
   Calibration: Per manufacturer's guidelines; standard curves
   Equipment: Carbon analyzer (Oceanography International,
     Model 524C, Oceanography International Corp., College
     Station, Tex.)
   Reference: EPA, 1974
   Sample Handling: Stored at 4 deg. C prior to analyses. TC ana-
     lyzed within 72 hr of sample collection. TOC and DOC ampulized
     and digested within 72 hr of sample collection. Filtered on day
     of collection. Analyses performed within 2 weeks.

Phosphorus

1. Total Phosphorus (TP)
   Method: Sulfuric acid/persulfate oxidation digestion; automated
     ascorbic acid colorimetric method

2. Total Soluble Phosphorus (TSP)
   Method: Sulfuric acid/persulfate oxidation digestion on sample
     filtered through 0.45-μ membrane filter; automated ascorbic
     acid colorimetric method

3. Soluble Reactive Phosphorus (SRP)
   Method: Automated ascorbic acid colorimetric method after
     filtration through a 0.45-μ membrane filter
   Detection Limits: 0.005 mg P/l (dependent upon range used in
     analyses)
   Calibration: Standard curves at beginning and end of each batch
     of samples
   Equipment: Auto Analyzers (Technicon Auto Analyzer II, Technicon
     Instruments Corp., Tarrytown, N. Y.)
   Reference: APHA, 1980
   Sample Handling: Stored at 4 deg. C prior to analyses, filtered
     day of collection. Anoxic samples filtered in field and held
     anoxic in syringes. Digestion on day of collection. SRP ana-
     lyzed within 48 hr of collection. TP and TSP analyzed within
     72 hr of collection.

Nitrogen

1. Total Nitrogen (TN)
   Method: Sulfuric acid/persulfate oxidation digestion; DeVarda's alloy reduction (Raveh and Avnimelech, 1979); automated phenol-hypochlorite colorimetric method

2. Total Soluble Nitrogen (TSN)
   Method: Same as above except sample was filtered through a 0.45-µ membrane filter prior to digestion

3. Ammonium-Nitrogen (NH_4N)
   Method: Automated phenol-hypochlorite colorimetric method

4. Nitrate-Nitrite Nitrogen (NO_3NO_2N)
   Method: Automated cadmium reduction colorimetric method sample filtered through a 0.45-µ membrane filter prior to analysis
   Detection Limits: 0.02 mg N/1 for TN, TSN, and NH_4N, 0.04 mg N/1 for NO_3NO_2N (dependent upon range used in analysis)
   Calibration: Standard curves at beginning and end of each batch of samples
   Equipment: Autoanalyzers (Techincon AutoAnalyzer II, Techincon Instrument Corp., Tarrytown, N. Y.)
   Reference: APHA, 1980
   Sample Handling: Stored at 4 deg. C prior to analyses, filtered day of collection. Anoxic samples filtered in field and held anoxic in syringes. Digestion on day of collection. DeVarda's alloy added 16 to 20 hr prior to analyses. NH_4N and NO_3NO_2N analyzed within 48 hr of sample collection. TN and TSN analyzed within 96 hr of sample collection.
   Comments: TN and TSN analyses performed on samples digested for TP and TSP, respectively (i.e., one digestion for both elements)

Sulfide

Method: Potentiometric
Detection Limit: 0.1 mg/1
Calibration: Standard curve
Equipment: Ion selective electrode and electronic voltmeter (Orion Research Ionanalyzer Model 901, Orion Research, Inc., Cambridge, Mass.)
Reference: Instruction Manual, Orion Research, 1980
Sample Handling: Collected with anoxic techniques in syringes and dispensed into buffer solution. Analyzed within 48 hr.

Sulfate

Method: Turbidimetric (nephelometric)
Detection Limit: 1.0 mg/1
Calibration: Standard curve, instrument calibrated per manufacturer
Equipment: Hach Model 2100A Turbidimeter (Hach Instruments, Ames, Iowa)
Reference: APHA, 1980
Sample Handling: Stored at 4 deg. C prior to analysis. Analyzed within 48 hr of sample collection.

Chloride

Method: Potentiometric
Detection Limit: 1.0 mg/l
Calibration: Standard curve
Equipment: Ion selective electrode and electronic voltmeter (Orion Research
  Ionanalyzer Model 901, Orion Research, Inc., Cambridge, Mass.)
Reference: APHA, 1980
Sample Handling: Stored at 4 deg. C prior to analysis. Analyzed within
  48 hr of sample collection.

Metals

1. Total Iron, Manganese, Calcium, Potassium, Magnesium, Sodium
    Method: Hydrochloric/nitric acid reflux digestion, atomic
      absorption spectrophotometry
    Detection Limits: 0.05 mg/l for iron and manganese, 0.1 mg/l for
      calcium, potassium, magnesium, and sodium

2. Dissolved Iron and Manganese (filtered through 0.1-μ membrane
    filter)
    Method: Filtered through a 0.1-μ membrane filter, atomic
      absorption spectrophotometry
    Detection Limit: 0.05 mg/l

3. Arsenic, Cadmium, Chromium, Copper, Lead, Nickel, Zinc
    Method: Atomic absorption spectrophotometry, graphite furnace
    Detection Limit: 0.0005 mg/l for lead; 0.001 mg/l for zinc, copper,
      arsenic, and nickel; 0.0005 mg/l for cadmium and chromium

4. Mercury
    Method: Atomic absorption spectrophotometry, cold-vapor technique
    Detection Limit: 0.001 mg/l
    Calibration: Standard curve
    Equipment: Atomic absorption spectrophotometer (Perkin-Elmer
      Model 4000); graphite furnace (Perkin-Elmer HGA-400); hydride
      generator (Perkin-Elmer MHS-10, Norwalk, Conn.)
    Reference: APHA, 1980
    Sample Handling: Filtered through 0.1-μ membrane filter day of
      collection. In-situ filtration of anoxic samples. All samples
      preserved with nitric acid to <pH 2. Digestion within 48 hr of
      sample collection. Dissolved variables analyzed within 72 hr of
      sample collection. Digested total variables analyzed within
      2 weeks of sample collection. Additional metals (i.e., C and D)
      analyzed within 3 months. Capabilities for these analyses are
      now available onsite and will reduce analysis time to within
      2 to 3 weeks.

Chlorophyll a

Method: Acetone extract; spectrophotometric
Calibration: Per manufacturer's guidelines
Equipment: Spectrophotometer (Perkin-Elmer, Model Lambda 3, Perkin-Elmer,
  Norwalk, Conn.)
Reference: APHA, 1980